EVALUATION OF ONSITE WASTEWATER TREATMENT AND DISPOSAL: VOLUME I

DETERMINATION OF GROUNDWATER CONTAMINATION AND DEMONSTRATION OF ALTERNATIVE TECHNOLOGIES

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Determination of Ground Water Contamination
From On-Site Wastewater Treatment and Disposal Systems,
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Executive Summary

Rapid development in unsewered areas of Bernalillo County has resulted in an increased threat to the ground water quality underlying these areas due to reliance of on-site systems for treating and disposing wastewater. Conventional on-site wastewater treatment and disposal systems consist of a septic tank which will remove most solids, and oils and grease. Septic tanks produce a very poor quality effluent which is discharged to the underlying soils through an absorption or leach field. The absorption field is intended solely to provide disposal of wastewater to the soil and is not designed to provide additional treatment.

Previous studies of ground water contamination from on-site systems in Bernalillo County have relied upon water samples collected from domestic wells. Water quality data from domestic wells is often of questionable value because well construction details are not known and these wells were not designed to provide water samples from the top of the water table, and thus the characteristics may not accurately reflect the quality of the region of the aquifer most strongly impacted by on-site disposal systems. The objective of this investigation was to determine ground water contamination from on-site systems through a combination of field monitoring, vadose zone sampling and numerical modeling studies.

The ground water monitoring program focused on areas of the county experiencing the most rapid growth and where sewer systems are not likely to be installed for many years. These are the Sandia Heights-North Albuquerque Acres area west of the Sandia Mountains and the area of Bernalillo County east of the Sandia and Monzano mountains which is referred to as the East Mountain area. An extensive search of hydrogeologic data and residential characteristics of the county identified five candidate areas for a ground water monitoring program: 1) Sandia Heights at the western base of the Sandia Mountains, and the East Mountain subdivisions of 2) Pinon Ridge, 3) Sierra Vista, 4) Sandia and 5) Sandia Knolls. Down gradient monitoring wells were installed in each of these areas except Sandia Knolls where drilling produced a dry hole. Ground water samples were collected from these wells quarterly for one year. These samples were analyzed for a variety of parameters to give information on the general water quality and the oxidation-reduction environment of the aquifer. Analyses were performed for major anions and

cations, iron, dissolved oxygen, nitrogen species, total organic carbon, and indicators of bacterial contamination. In addition, the United States Geological Survey (USGS) collected samples from most of these wells and subjected them to an extensive analysis for a suite of 66 organic compounds associated with various types of human activities and wastes. The monitoring program found concentrations of nitrates up to 3.5 mg N/L as NO_3^- in the Sandia Heights wells, and 3.5 mg N/L as NO_3^- in one of the Sandia Park wells. Evidence of reducing conditions were found in the other East Mountain wells which is significant because reducing conditions will prevent formation of nitrates in subsurface environments. This evidence included low dissolved oxygen concentrations (< 1.0 mg/L), elevated iron (.3 to .6 mg/L), and total organic carbon concentrations greater than 2.0 mg/L. The USGS sampling and analyses for organic constituents found low but detectable concentrations of a number of anthropogenic organic compounds in the East Mountain wells, which provides further evidence of ground water contamination from on-site wastewater systems in this area.

A vadose zone sampling program was conducted to investigate the transport and transformation of wastewater contaminants in the unsaturated soils beneath an operating septic tank-absorption field system. A site was identified in North Albuquerque Acres just west of the Sandia Heights ground water monitoring area which located in an area of unconsolidated alluvial deposits of the Upper Santa Fe Formation, with a depth to ground water in excess of 500 ft. Five holes were drilled to a depth of 65 ft with a hollow stem auger, and soil samples were collected with a split spoon sampler at five ft intervals. These samples were analyzed for gravimetric water content, nitrogen species, chemical oxygen demand (COD), chloride, sulfate and bacterial species. In addition, a bromide tracer was added to the wastewater two weeks before the sampling program to assist in evaluating the rate of vertical flow of water through the unsaturated soils. The investigation found rapid water movement through the soil column with bromide being detected throughout the 65 ft deep hole closest to the leach field. Nitrification of septic tank effluent occurred in less than 10 ft below the drain field and was manifested by pore water nitrate concentration of 75 mg N/L. Pore water COD concentrations decrease from 5000 mg/L immediately beneath the leach field to approximately 1000 mg/L at 65 ft depth. The limited data from this study appear to show decreasing nitrate and COD concentrations with depth, suggesting possible occurrence of denitrification in the soil column, although the general subsurface

environment shows evidence of aerobic conditions. Additional work is needed to confirm nitrogen disappearance in the vadose zone.

A ground water flow model was calibrated and used to calculate the flow regime needed to develop a non-reactive contaminant transport model for the aquifer beneath the Sandia Heights subdivision to assist in interpreting the data from the ground water monitoring program. The flow model used hydraulic parameters identified by the USGS for its regional model of the Albuquerque Basin, including a mountain front recharge rate of 12.2 ft^3/s along the 10 mile length of the western face of the Sandia mountains. Two north trending faults pass through the study area which were modeled as low hydraulic conductivity zones in the flow model. The influence of on-site disposal systems on the aquifer hydraulics was simulated by addition of 0.422 acre-ft/yr of infiltration for each system. Nitrates were assumed to be non-reactive once they reached the water table, which allowed the impact of absorption field leachate on water quality to be modeled by adding 23.8 mg/L of a conservative tracer to the infiltration flow. The contaminant transport model predicted a maximum nitrate concentration would occur after approximately 50 years of wastewater disposal through the on-site systems, and the peak concentration would be about 6. mg N/L. After 30 years of waste disposal would produce a peak nitrate concentration of about 5 mg N/L which is close to the value of 3.5 mg N/L measured during the monitoring program. A sensitivity analysis was performed which investigated the effects of lot size, mountain front recharge, and leachate nitrate concentration on the maximum ground water nitrate concentration. The model was most sensitive to the nitrate concentration in the wastewater, with lot size being the second most important parameter. There is an approximately linear relationship between the maximum nitrate concentration and housing density and nitrate loading from the disposal systems. In other words, doubling the density of the subdivision will raise the maximum nitrate concentration from 6 mg N/L to approximately 11 mg N/L, and increasing the nitrate concentration in the effluent from 23.8 mg N/L to 48 mg N/L will increase the maximum ground water nitrate concentration to 12 mg N/L. There is less dependence on the mountain front recharge rate.

The results of this investigation confirmed that on-site wastewater treatment and disposal systems have impacted ground water quality in Bernalillo County as has been found in previous investigations. In the Sandia Heights subdivision, this contamination appears in the form of

elevated nitrate concentrations. The modeling results suggest that since the subdivision is somewhat less than 30 years old, the nitrate concentrations in the underlying ground water may be increasing with time and may eventually approach or exceed the State ground water standard. The hydrogeology of the East Mountain area is much more complicated and it appears that in many locations subsurface conditions are not suitable for formation of an extensive nitrate problem. However, it appears that other types of ground water quality problems may be present due to widespread utilization of on-site systems. These include unpleasant taste and odors associated with anaerobic conditions, possible bacterial contamination of the water bearing strata, and the presence of synthetic organic contaminants in the underlying aquifers.

Perhaps the most significant conclusion of this investigation is a reaffirmation of the diversity and extraordinary complexity of the hydrogeologic environment which occurs in Bernalillo County. Due to this variability and complexity it is important to recognize that the types of ground water problems resulting from on-site wastewater treatment and disposal will also vary. Therefore, it is recommended that local and state ground water protection regulations should not focus on a single water quality criteria, but rather they should address the protection of the overall quality of the ground water resource. In particular, past regulatory practice has been directed primarily towards prevention of nitrate contamination of ground water. This is due in large part to the fact that high nitrate concentrations can often be unequivocally attributed to on-site disposal systems, and because this parameter is cheap and easy to measure. In contrast, measurement of synthetic organic compounds in a ground water sample requires special sampling and analytical procedures that are difficult and very costly. This study has demonstrated that subsurface environmental conditions in many areas of Bernalillo County are not suitable for generation of nitrates, yet that doesn't mean that ground water contamination has not occurred or will not occur in the future.

Continued monitoring of ground water quality in the County is imperative, particularly in light of model predictions which suggest further deterioration of water quality. This monitoring should include sampling and analysis for a spectrum of contaminants including both inorganics organic constituents as well as indicators of microbial quality. Finally, in light of the findings of this study that ground water contamination has occurred in a variety of hydrogeologic environments throughout the County, development and implementation of stringent ground water

quality protection measures is needed to protect this vital resource for current and future generations.

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INTRODUCTION

The past 30 years has found ever increasing and rapid development in rural parts of New Mexico, especially in Bernalillo County. For example, since 1980 the annual population growth in the East Mountain area of Bernalillo County has consistently averaged approximately 5% (Herrington, 1998). Growth in this area in recent years continues unabated, and the population is expected to double to 30,000 by the year 2020.

Along with population growth in rural areas has come increasing recognition of the vulnerability of groundwater systems to contamination from surface and near surface activities, particularly use of conventional on-site wastewater treatment and disposal systems. For purposes of this project conventional systems are assumed to be septic tank-absorption field systems. These systems consist of a septic tank to allow separation of solids and floating materials (i.e. grease and oil), followed by disposal of the settled wastewater into the ground through an absorption field. A description of these and other on-site wastewater treatment and disposal alternatives has been published by the EPA (USEPA, 1980).

1.1.   Ground-Water Protection Policy and Action Plan

Recognizing the problems associated with a variety of threats to groundwater resources in the Albuquerque area, the Albuquerque City Council and the Bernalillo County Commission funded development of the "Albuquerque/Bernalillo County Ground-Water Protection Policy and Action Plan" (GPPAP). This plan was developed over a period of almost 6 years and included publication of 12 technical reports dealing with threats to the ground-water resource and possible protection strategies. Details of the GPPAP and its development are contained in the report entitled "Ground-Water Protection Policy and Action Plan" prepared by the Policy Coordinating Committee (1995). The GPPAP was adopted by the County in November 1993 and by the City in August 1994, and identified 3 specific goals:

1. Protect the ground-water resource,
2. Find and clean up the contaminated ground water,

3. Promote the coordinated protection and prudent use of the ground-water resource throughout the region.

The GPPAP proposed that 6 policies be adopted to achieve the goals of the GPPAP. The first policy is the most relevant to the proposed work described in this document. It is: "POLICY A: The City and County shall prohibit or control the releases of substances having the potential to degrade the ground-water quality." In discussing this policy, the GPPAP identifies the highest priority threats to groundwater quality as:

- Underground Storage Tanks;
- Hazardous materials and waste storage facilities; and
- On-site liquid waste disposal systems.

The GPPAP contained a number of Action Plans which were intended to achieve its objectives. Two action plans were specifically identified to address the threats posed by on-site waste disposal systems: (1) continuing and expanding environmental monitoring programs to document the groundwater problems associated with on-site treatment and disposal systems, and (2) demonstrate the performance of alternative on-site systems and other liquid waste disposal options which are capable of meeting State ground water standards (New Mexico Water Quality Control Commission, 1996).

1.2. Study Objectives

The objectives of the overall study described in this report were developed by referencing the descriptions of the two Action Plans contained on pages 66 and 67 of the GPPAP (Policy Coordinating Committee, 1995). The two principal objectives of the Action Plans are:

- Determine the extent of groundwater degradation in areas served by on-site wastewater disposal systems,
- Demonstrate the ability of alternative on-site systems to remove conventional wastewater pollutants, particularly nitrogen contaminants.

Several intermediate tasks were specifically identified to accomplish each of these objectives in the GPPAP. Specific tasks which required completion in order to document the impact of on-site wastewater systems on the quality of underlying ground water included:

- Document ground-water degradation from existing on-site liquid waste disposal systems.
- Determine the potential for natural processes to attenuate contamination from on-site systems.
- Comment on the planning tools used to develop the lot-size and density limitation guidelines.
- Develop criteria that can be used to determine lot-size guidelines for household liquid waste disposal alternatives.

Intermediate tasks which were required to demonstrate the performance of alternative on-site wastewater treatment and disposal systems included:

- Identify alternative on-site wastewater treatment and disposal technologies which show the capability of meeting State ground water standards
- Demonstrate performance of alternative on-site wastewater treatment and disposal systems.
- Develop performance criteria for alternative on-site systems.
- Identify operations and maintenance requirements for alternative on-site systems.
- Identify changes to regulations (such as the Uniform Plumbing Code or State Liquid Waste Disposal Regulations (20 NMAC 7.3)) to assist City and/or County staff in their evaluation and approval of alternative on-site systems.

This report describes an investigation conducted by the Waste-management Education and Research Consortium (WERC), and by researchers and students at the University of New Mexico and New Mexico State University of the problems associated resulting from on-site

wastewater treatment and disposal systems in Bernalillo County and possible technologies for providing improved wastewater treatment. The report has two main parts. Volume I describes work conducted to identify impacts on ground water quality from on-site wastewater treatment and disposal systems in unsewered parts of the county. Volume II describes an investigation undertaken to evaluate the performance of alternative on-site wastewater treatment systems. A brief description of each phase of the study is presented in the following sections.

1.3. Phase I Objectives: Determination of Ground Water Contamination

Phase I of the project consisted of a series of field and modeling investigations which were designed to detect ground water contamination from conventional on-site wastewater treatment and disposal systems. The principal tasks associated with this phase consisted of the following:

- Collect and assimilate all available information on ground water hydrology and ground water quality in Bernalillo County with an emphasis on the northeast heights on the west side of the Sandia Mountains, and the East Mountain area;

- Perform a scoping study to identify unsewered parts of the County with a high density of development and significant potential for ground water contamination;

- Design and implement a ground water monitoring program, including drilling and sampling of monitoring wells, which would provide information on ground water quality in select areas;

- Conduct a vadose zone sampling program to characterize contaminant transformations and transport in deep unsaturated geologic formations;

- Conduct a modeling study to assist in understanding potential ground water contamination from on-site wastewater treatment and disposal.

1.4. Phase II. Objectives: Demonstration of Alternative Technologies

The principal objective of this phase of the study was to demonstrate the ability of alternative on-site wastewater treatment systems to remove wastewater pollutants (including organics, suspended solids, pathogenic organisms, and nitrogen compounds) in order to permit

on-site discharge of this wastewater without causing deterioration of underlying ground water quality. This phase of the project had two main aspects. First, information was collected on the availability and performance of existing alternative on-site treatment systems. A selection criteria was used to select the most promising systems and potential demonstration sites were located. The second stage of the study involved installing these systems and operating and monitoring them to develop information regarding their cost, performance, reliability, and operations and maintenance requirements. Recommendations for evaluating future alternative systems were developed. This study, its conclusions and recommendations are contained in Volume II of this report.

GROUND WATER CONTAMINATION FROM ON-SITE SYSTEMS

1.5.   Theoretical Considerations

The contaminants found in domestic wastewater can be divided into three categories: physical pollutants, chemical pollutants, and microbiological pollutants (Metcalf and Eddy, Inc., 1990). The principal physical constituents of domestic wastewater are suspended solids which are readily removed by filtration and sedimentation in the soil column immediately beneath an absorption field trench. The conventional chemical contaminants of concern in domestic wastewater include organic material consisting of a large variety of mostly biodegradable compounds which are combined and reported as 5-day biochemical oxygen demand (BOD_5), nitrogenous compounds (principally ammonia and organic nitrogen species), and phosphates.

Microbial contaminants include viruses, bacteria, protozoans and other microorganisms. Microbial contamination from wastewater is almost always characterized by measuring the presence and/or concentration of indicator organisms. Organisms which are usually used as indications of human waste include total coliform bacteria, fecal coliform bacteria and streptococcus bacteria. Because they are particulate in nature and are usually filtered by a few feet of subsurface soils, microbial contaminants seldom constitute a regional ground water problem, hence they will not be considered further in this discussion. However, because microbial contamination can cause problems in fractured formations similar to those which are found at some locations in the East Mountain area, coliform bacteria were analyzed in ground water samples collected during this project. Table 0-1 contains a list of typical contaminants from on-site systems, their mass loading (i.e. mass of contaminant generated per person per day), and typical concentration range (USEPA, 1980).

Conventional onsite wastewater treatment and disposal is accomplished through the use of septic tank-absorption field systems (USEPA, 1980). In these systems a septic tank provides the following functions: 1) allows solid material to settle; 2) allows oil, grease and other floatables to rise to the surface; and 3) provides volume for accumulation of the solids and floatables, which allows for some anaerobic degradation of these compounds. The absorption field (also known as the drain or leach field) is designed solely to optimize disposal of wastewater from the septic tank. It is important to recognize that conventional onsite wastewater

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treatment and disposal systems are not designed to provide any treatment function other than removal of solids and floatables.

Constituent	Mass Loading (g/person-day)	Concentration (mg/L)
Total Solids	115-170	680-1000
Volatile Solids	65-85	380-500
Suspended Solids	35-50	200-290
Volatile Suspended Solids	25-40	150-240
BOD_5	35-50	200-290
COD	115-125	680-730
Total N	6-17	35-100
NH_3 (as N)	1-3	6-18
TKN (as N)	5.5-18.0	11.3
NO_2^- & NO_3^- (as N)	<1	<1
Total P	3-5	18-29
PO_4^{3-}	1-4	6-24
SO_4^{2-}	4-19	11.
Total Coliforms (org./100mL)	-	10^{10}-10^{12}
Fecal Coliforms (org./100mL)	-	10^8-10^{10}

Table 0-1. Concentrations of contaminants in effluents from grey water and black water septic tanks (USEPA, 1980).

Reference to Table 0-1 illustrates a potential point of confusion that is clarified here. Because there are many forms of nitrogen compounds in water it is common practice to report them in terms of the mass of nitrogen per liter rather than as the mass of say nitrate, nitrite, or ammonium. Therefore, 10 mg N/L as NH_4^+ (molecular weight 18 g/mole) may be stoichiometrically oxidized to produce 10 mg N/L as NO_3^- (molecular weight 62 g/mole). Furthermore, 10 mg N/L as NO_3^- corresponds to 44.3 mg NO_3^-/L. Accordingly, the original U.S. Public Health Service drinking water standard for nitrate was 45 mg NO_3^-/L. The convention of

reporting all nitrogen concentrations as mg N/L regardless of their species is followed throughout this report.

The principal chemical transformations which occur beneath a leach field are those associated with microbial degradation of organic constituents and nitrogen species. This process is sometimes referred to as mineralization. In subsurface environments there will be a sequence of microbial transformations depending on the available energy source (also referred to as substrate or electron donor) and terminal electron acceptor. Generally, reactions which provide the most energy to the microorganisms proceed faster and will occur first, while reactions which provide less energy occur later. Domestic wastewater contains high concentrations of a wide variety of readily degradable constituents, therefore, the microbial growth sequence in the soil beneath an absorption field is determined by the available electron acceptor. Common electron acceptors and the free energy of the half reactions are listed in Table 0-2. The second column represents the standard Gibbs Free Energy of the half reactions (1 M concentration of all compounds, T = 298 K, and P = 1 atm), while the third column in this table represents the actual Gibbs Free Energy expected under conditions similar to those in the subsurface environment (1 mg/L concentration of selected compounds, pH = 7, T = 298 K, and partial pressure of all gases = 1 atm). This table gives a semi-quantitative comparison of the energy available to microbial populations in the presence of different common terminal electron acceptors. Note that all of these reactions are written as reductions and reflect the fact that various microorganisms use these compounds as their terminal electron acceptor in microbially mediated oxidation-reduction reactions.

Reaction	$\Delta G°$ (kJ/mol)	ΔG (kJ/mol)	
$1/4\ O_{2(g)} + H^+ + e^- = 1/2\ H_2O$	-118.56	-78.6	(2.1)
$1/5\ NO_3^- + 6/5\ H^+ + e^- = 1/10\ N_{2(g)} + 3/5\ H_2O$	-120.52	-67.8	(2.2)
$UO_2^{2+} + 2\ e^- = UO_{2(s)}$	-39.1-	-23.6	(2.3)
$Fe(OH)_{3(s)} + 3\ H^+ + e^- = Fe^{2+} + 3\ H_2O$	-93.79	-1.0	(2.4)
$1/8\ SO_4^{2-} + 9/8\ H^+ + e^- = 1/8\ HS^- + 1/2\ H_2O$	-23.99	20.9	(2.5)
$1/8\ CO_{2(g)} + H^+ + e^- = 1/8\ CH_{4(g)} + 1/4\ H_2O$	-14.27	25.7	(2.6)

Table 0-2. Gibbs Free Energy for important microbial half reactions. ($\Delta G°$ is the standard free energy for the reaction and ΔG represents the free energy expected under typical environmental conditions with pressure of all gases = 1 atm, and the concentrations of NO_3^-, UO_2^{2+}, and Fe^{2+} = 1 mg/L (adapted from Stumm and Morgan, 1996).)

The most common substrates (electron donor) in wastewater include a wide variety of organic compounds (represented here as CH_2O) and nitrogen species. Organic compounds in domestic wastewater are readily oxidized under both aerobic and anaerobic conditions. The half reaction for this oxidation is shown by equation (2.7):

$$1/4\ CH2O + 1/4\ H2O = 1/4\ CO2 + H+ + e- \qquad\qquad (2.7)$$
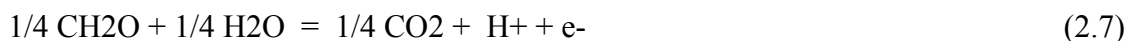
Classes of organisms capable of mediating this reaction while using O_2 (aerobic organisms), NO_3^- (denitrifying organisms), oxidized metals such as iron(III) and uranium(VI) (metal reducing bacteria), SO_4^{2-} (sulfate reducing bacteria), and CO_2 (methanogenic bacteria) as their terminal electron acceptor are well known and believed to be ubiquitous in subsurface soils (Chappelle, 1993).

The principal transformations of nitrogen compounds include conversion of organic nitrogen species to ammonia (NH_3) followed by oxidation to nitrate (NO_3^-). The hydrolysis reaction is sometimes referred to as ammonification and occurs through breakdown of organic tissues containing amino acids, and hydrolysis of urea. These are rapid reactions and nearly 100% of organic nitrogen in wastewater is readily converted to NH_3 (Kadlec and Knight, 1996). Oxidation of NH_3^+, known as nitrification, is a sequential process mediated by facultative autotrophic bacteria under aerobic conditions. The overall half reaction is:

$$1/8 \text{ NH}_4^+ + 3/8 \text{ H}_2\text{O} \;=\; 1/8 \text{ NO}_3^- + 5/4 \text{ H}^+ + \text{e}^- \tag{2.8}$$
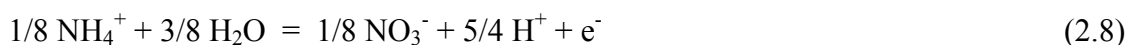
It is important to recognize that the nitrification reaction requires the presence of O_2. Coupling reaction (2.8), oxidation of NH_4^+, with the half reaction for reduction of O_2, reaction (2.1), yields the overall oxidation of NH_4^+:

$$\text{NH}_4^+ + 2 \text{ O}_{2(g)} \;=\; \text{NO}_3^- + 2 \text{ H}^+ + \text{H}_2\text{O} \tag{2.9}$$

The nitrification reaction is slower than reaction (2.7), it requires the presence of a high concentration of oxygen, and it is strongly temperature dependent (Keeney, 1986). The nitrification reaction requires dissolved oxygen concentrations greater than 1 mg/L and temperatures greater than 5°C. Coarse textured, strongly aggregated soils with large depth to ground water favor nitrification, while fine textured, poorly aggregated soils and shallow depth to ground water limit this reaction (Reneau et al., 1989).

Microbial denitrification is the reduction of nitrate by heterotrophic organisms which use organic carbon as their carbon source and NO_3^- as their electron acceptor. Denitrification is a sequential process catalyzed by a number of different organisms and proceeding in the sequence: $NO_3^- \rightarrow NO_2^- \rightarrow N_2O \rightarrow N_2$. The overall reaction can be represented as follows, where CH_2O again represents a generic organic carbon source:

$$4 \text{ NO}_3^- + 5 \text{ CH}_2\text{O} + 4 \text{ H}^+ \;=\; 2 \text{ N}_{2(g)} + 5 \text{ CO}_{2(g)} + 7 \text{ H}_2\text{O} \tag{2.10}$$
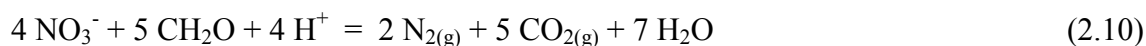
Conditions necessary for this reaction to occur include the presence of an organic carbon source and the absence of O_2. While many denitrifying organisms are facultative and therefore can grow in the presence of O_2, the organism recovers more energy from the O_2 reduction (half reaction (2.1)), therefore O_2 is used preferentially by the organisms if present (Chappelle, 1992).

One of the main concerns of on-site waste disposal is the accumulation of nitrates in upper ground water zones. The national drinking water standard of 10 mg N/L as NO_3^- for domestic water supplies was enacted because of the potential risk of methemoglobinemia (blue-

baby syndrome) to bottle-fed infants, and in view of the absence of substantiated physiological effects at nitrate concentrations below this level (US EPA, 1986).

There have been a large number of studies of potential ground water pollution from conventional on-site wastewater disposal systems. Some of the most relevant studies include those by Piluk and Howe (1989), Reneau et al., (1989), Robertson and Cherry (1992), Robertson et al. (1992), and Starr and Sawhney (1980). Recently Wilhelm et al., (1994) published a review paper which proposed a conceptual model of the biological and chemical transformations which occur in a conventional septic tank-absorption field system. According to this model, the evolution of waste water is driven by the microbially catalyzed redox reactions involving organic C and N in wastewater, which can occur in as many as three different redox zones.

The first zone consists primarily of the septic tank and represents the initial stages of the wastewater treatment. Distribution boxes evenly divide the flow to the perforated drain lines buried in the absorption field. From there the effluent infiltrates through the soil where it is subject to purification through a variety of processes including filtration, sorption, and biodegradation.

The second zone in this conceptual model consists of the soil beneath the absorption field where transport of gases by diffusion through the unsaturated soil provides O_2 for aerobic oxidation of organic C and NH_4^+(nitrification). Note that the nitrification reaction (reaction (2.9)) reduces the alkalinity and pH of the water. The main product and constituent of concern in ground water contamination, in this zone is the NO_3^- formed by the oxidation of NH_4^+. The concern over ground water contamination from nitrate can be explained by the fact that it is a stable anionic species, which readily passes through soil along with percolating effluent and other recharge waters until it reaches the water table (Virginia, 1997). According to previous studies virtually complete nitrification of ammonium-nitrogen has been found to occur in the unsaturated zone in well-aerated soil below septic tank disposal fields (Hantzsche, et. al., 1992).

Autotrophic nitrification predominates; however, a variety of heterotrophic microorganisms contribute to some degree to the nitrifying conversion process. Heterotrophic nitrification is only likely to be of importance where autotrophic organisms are inactive (due perhaps to low pH). Most nitrification in aerobic soils is attributable to two chemolithotrophic

bacteria, which utilize inorganic NH_4^+ and NO_2^- as their energy source (electron donors) (Payne, 1981). The first are *nitrosomonas*, which oxidizes NH_4^+ to the intermediate product NO_2^-. The second genera of bacteria are *nitrobacter*, which oxidizes NO_2^- to NO_3^-.

The rates at which the nitrification process will take place are dependent upon available NH_4^+ or NO_2^-, pH, temperature, O_2 and soil moisture. Previous studies have shown that the most important factor controlling nitrification rates is the availability of O_2 nitrifiers. Nitrifiers are obligate aerobes that use oxygen as a terminal electron acceptor (Prosser, 1986). Therefore, these microbes are most efficient in aerobic, well-drained soils; which are usually the conditions found beneath a septic tank leach field.

The optimum pH for nitrification in soils is from 6.6 to 8.0. At pH values greater than 8.5, nitrification may be inhibited due to NH_3 toxicity. Recent studies have indicated that rates of nitrification also rapidly decline below pH 6 (Kadlec and Knight, 1996). The oxidation of NH_4^+ to NO_2^- results in an increase in acidity within the system. Decreases in pH, however, may have only minimal effects on nitrification in some soil systems, because of the buffering capacity of natural soils, and also the fact that several strains of nitrifying organisms are adapted to acid soil conditions (pH values as low as 4). Heterotrophic nitrifiers may also grow under acidic pH conditions.

The temperature of the soil has a large influence over the rate at which the bacteria are able to convert the NH_4^+ to NO_3^-. Laboratory studies suggest that the optimal temperature for nitrification is between 30 and 35° C, however, nitrifying organisms grow well under mesophilic conditions as well (temperatures in the range of 5 to 20 °C) (Black, 1968). The USEPA (1993) has published data which suggest that below temperatures of 5 °C nitrification reactions slow to the point of becoming insignificant. Note however, that subsurface soils in the temperate regions of the U.S. have temperatures much greater than this. Soil temperatures at a depth of greater than 5 m in Albuquerque, NM are approximately 15 °C.

Nitrifying bacteria are sensitive organisms and extremely susceptible to the wide variety of inhibitors mentioned above. Summaries of the main parameters, which can greatly affect the rate of nitrification in a soil system, are as follows:

- A variety of organic and inorganic agents can inhibit the growth and action of nitrifying and denitrifying organisms.

- The pH of the soil-water system is important. The optimum pH is in the range of 7.5 to 8.6, but systems acclimated to lower pH conditions also can successfully nitrify.

- Temperature exerts tremendous influence on the growth of nitrifying bacteria, although at depths greater than a few meters the soil temperature is relatively constant.

- Dissolved oxygen concentrations of 1 mg/L are essential for nitrification. If DO levels drop below this value, O_2 becomes the limiting nutrient and nitrification slows or ceases.

In the third zone of the model proposed by Wilhelm et al. (1994), NO_3^- is reduced to N_2 by the anaerobic denitrification process. This process refers to the biological or chemical reduction of nitrate and nitrite to volatile gases, usually nitrous oxide and molecular nitrogen. Presently, denitrification is the only mechanism in the soil known to effect significant reduction of nitrate in percolating effluent (Wilhelm et al. 1994). The rate of denitrification is dependent upon the concentrations of NO_3^-, the presence of a suitable substrate for microbial growth, and the lack of O_2. Environmental factors including pH, temperature, and moisture content of the soil will also affect the rate of denitrification. In the absence of O_2, the principal limiting factor affecting the rate of denitrification, is the concentration of soluble carbon. Wilhelm et al. (1994) have suggested that denitrification will not take place beneath a septic tank absorption field due to a lack of labile organic C within the soil system.

The optimal pH for denitrification varies depending upon the soil. Russell (1973) and Alexander (1977) reported that the highest rates of denitrification occurred in soils with pH values ranging from 8-8.6, but in some soils the reaction may be rapid at a pH of 4.7. Bremmer and Shaw (1956) examined relationships between soil pH and denitrification. Their results showed that soils with pH values of 3.6, 4.1 and 7.5 indicated minimal amounts of denitrification. Soils with pH values of 8.2 however, produced the highest amounts of denitrification. Stanford et al. (1975a) also examined the relationship between soil pH and rates of denitrification and found no correlation between the two. A study by Fillery (1983) suggested that at high pH conditions, levels of soluble C increase. Therefore, increases in rates of denitrification may be a response to the additional C released at the higher pH.

Similar to nitrification, the rate of denitrification is also temperature dependent. Several studies have concluded that denitrification rates increase with temperature. In experiments with clay and sandy loams, denitrifying activity was minimal at 2-5° C but increased linearly with temperature up to 25° C. From 25° up to 60° C, activity continued to increase slightly, then dropped sharply to inactivity (Payne, 1981). Stanford et al. (1975b) showed similar results. A steady increase in activity occurred between 15 and 35° C, and a rapid decrease in rates occurred at temperatures below 10° C.

Moisture content of the soil exhibits a large influence over the rate of denitrification. Numerous studies have shown relationships between moisture content and denitrification rates. Freney et. al. (1979) observed a marked increase in rates of N_2O production as water was added to the soil (up to 62% saturation). In a similar study, Bremmer and Shaw (1956) observed minimal NO_3^- reduction when the moisture content was less than 60% of the water holding capacity of the soil. In a later study by Mahendrappa and Smith (1967), the effect of moisture content on denitrification was examined on four different soils (2 acid and 2 basic in batch studies). They noted that each soil exhibited a moisture level in which denitrification was optimized. The optimum moisture content was related to an equal distribution of nitrogen products and microbes, as well as dilution of toxic products produced.

Reference to Table 0-1 shows that septic tank effluents do not have significant concentrations of nitrate due to the strongly reducing conditions in effluent from septic tanks. Nitrification may occur in the unsaturated soil beneath the absorption field if aerobic conditions are present and there is an adequate amount of time for the nitrification reactions to proceed before the soil pore water reaches the water table. If subsurface environmental conditions are not favorable for denitrification, then the NO_3^- produced in the vadose zone will ultimately reach the water table and contaminate the ground water. In the conceptual model of the evolution of septic tank effluent, Wilhelm et al. (1994) propose that the most favorable soil conditions for denitrification are:

- the abundance of an organic carbon sustrate,
- high soil moisture content, and high soil pH
- absence of O_2
- high moisture content.

In another study, Stuart, et. al. (1995) suggested similar conditions for denitrification, however, it was noted that a low soil pH was needed. One explanation for the difference may be due to natural variation among denitrifying organisms. As in the nitrification process, if pH values are low, heterotrophic organisms will expedite the denitrification process; while at high pH values other organisms seem to thrive. Nevertheless, a well buffered soil with pH between 6.0 and 8.5 containing a degradable organic carbon source and anaerobic conditions would be expected to support an active culture of denitrifying organisms capable of reducing nitrate concentrations to below detectable levels.

The effective treatment of septic tank effluent is critical in order to protect ground water quality. Presently, the contaminant of primary concern is nitrate. Previous studies have proven that in most aerobic soils nitrate can move rapidly with minimum reduction away from the absorption field and into the ground water (Reneau, et al., 1989; Robertson and Cherry, 1992; Robertson, et al. 1991) The degree to which the soil can purify and filter wastewater is primarily dependent upon the physical characteristics of the soil. Physical properties such as soil structure, saturated hydraulic conductivity, and hydrologic relationships of water entering and moving through the soil directly effect travel times, and thus wastewater-soil interactions. In general, the slower the wastewater moves through the soil the more likely soil-wastewater interactions will occur leading to increased filtration and purification (USGS, 1993). Soils with coarse textures, strong structure, or under saturated conditions are the most conducive to nitrification reactions and subsequent transport of NO_3^-. In these cases dilution is the only form of effective nitrogen treatment between the absorption field and the ground water.

This discussion of the theoretical aspects of biologically mediated reactions in the subsurface environment can be used to develop a conceptual model of the chemical transformations in the soil beneath an onsite wastewater disposal system (Figure 2.1). This model illustrates that there is a sequence of zones, each characterized by a predominating biologically active redox couple. In contrast to the model proposed by Wilhelm et al. (1994), this model focuses on transport and transformations in the subsurface environment beneath an absorption field. The model is similar the generic model developed by Lovley et al. (1994) for

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the evolution of ground water quality as it flows from a zone of oxidizing conditions to reducing conditions.

Immediately beneath the absorption field a very high concentration of readily degradable organic compounds results in depletion of the highest energy electron acceptors including O_2 and NO_3^-. Most of the particulate material and much of the soluble organic associated with the septic tank effluent is trapped or adsorbed on the soil particles close to the drain field resulting in a very high concentrations of biochemical oxygen demand. Microbial consumption of O_2 exceeds the diffusive O_2 transfer rate, therefore, strongly reducing, anaerobic conditions prevail. Oxygen and NO_3^- provide the most energy to the microbial population, and therefore result in the most rapid microbial growth. They are rapidly consumed in this zone resulting in strongly reducing conditions. Anaerobic reactions will result in degradation of these carbonaceous materials with consequent production of Fe(II), Mn(II), HS^-, and CH_4. The organic carbon content will decrease with depth, so that eventually O_2 diffusion will be sufficient to maintain aerobic conditions. This will permit further mineralization of organic carbon as well as nitrification. The final reaction in this proposed sequence will be production of NO_3^- through the nitrification process. Although, as discussed above, microbial denitrification is possible, based on the conceptual model presented here, NO_3^- will remain in the soil and water column because the conditions for reduction of nitrate are not present. Specifically, the conditions preventing denitrification are: 1) O_2 is present and 2) all of the degradable organic carbon has been consumed in the upper soil horizons thus there is no energy source for nitrate reaction. The soil pore water chemistry will show evidence of high concentrations of CH_4, Fe(II), and HS^- corresponding to microbial use of CO_2, $Fe(OH)_{3(s)}$, and $SO_4^{2=}$ as terminal electron acceptors.

The conceptual model depicted in Figure 0-1 illustrates the transport and transformations associated with wastewater in a deep unsaturated soil column. Hydrogeologic conditions may preclude full development of this profile. The most common limitation is the presence of water table conditions near the surface so that the contaminants will reach ground water before nitrification can occur. Because O_2 transport through ground water via advection and diffusion is very low, production of nitrate by nitrification below the water will occur very slowly if at all. Anaerobic conditions will persist in the ground water which is likely to exhibit high concentrations of Fe(II), Mn(II), HS^- and CH_4. This model is consistent with the prevalence of

reducing conditions in the inner valley of the Rio Grande near Albuquerque, Las Cruces, and other communities along the river. In addition to depth to ground water, low permeability of the unsaturated soil may limit full development of the conditions shown in Figure 0-1. Under these conditions O_2 transport may not be adequate to sustain aerobic conditions at depth so that nitrification does not occur.
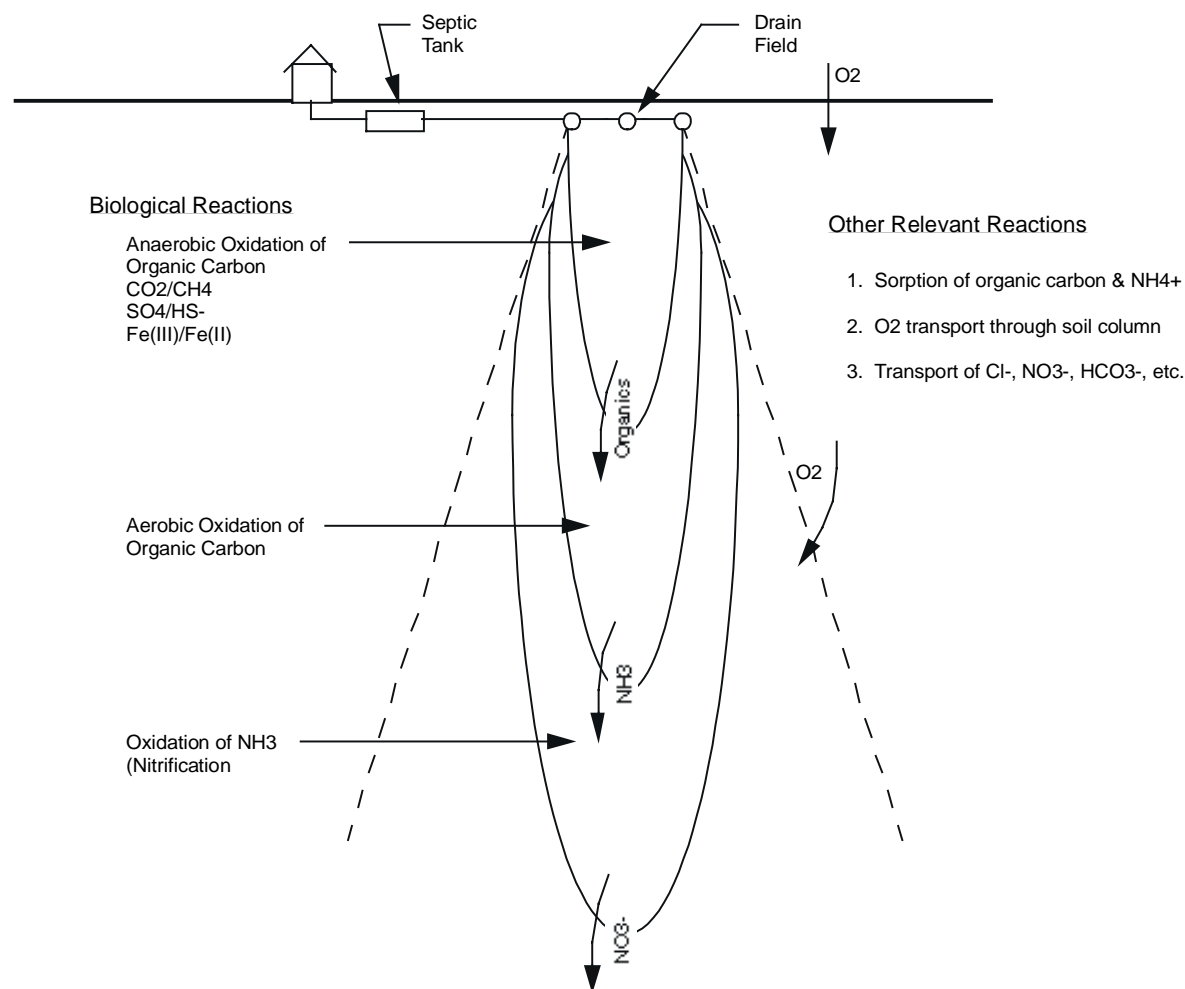
Figure 0-1. Conceptual model illustrating the transport and transformations of carbon and nitrogen compounds beneath a conventional onsite wastewater treatment and disposal system.

The cumulative effect of a high density of onsite wastewater treatment and disposal systems is illustrated in Figure 0-2. The contaminant plume expands with depth and if the systems are sufficiently close together, the plumes will co-mingle. This may result in the oxygen demand of the wastewater exceeding the O_2 transfer capacity of the soil causing the entire soil

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column to become anaerobic. The absence of O_2 will prevent generation of NO_3^- by nitrification. Nitrogen species would still be present, most likely as NH_4^+ sorbed to the soil, but NO_3^- concentrations would be limited by the rate of O_2 transfer through the soil column.

Limit of impact of individual drain field
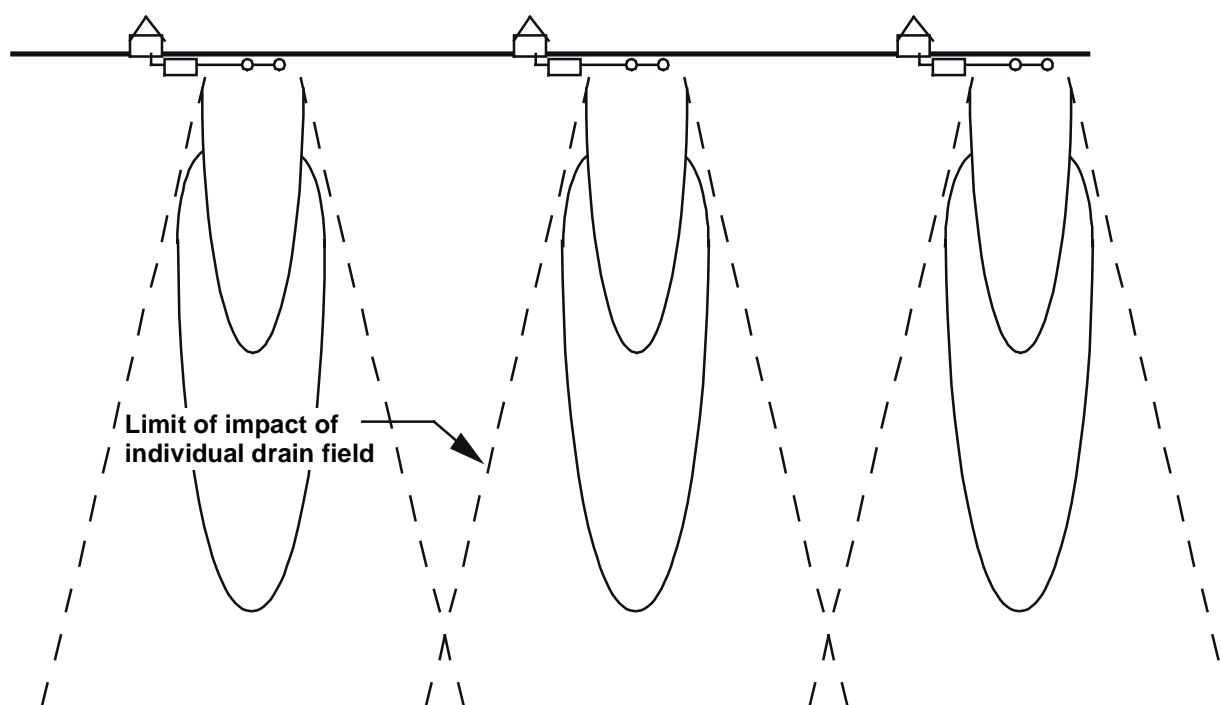
Figure 0-2. Conceptual model of the impact of a high density of onsite wastewater treatment and disposal systems on the subsurface environment.

Only one previous study was identified which investigated the impacts of on-site disposal systems in arid climates with large distances to the ground water. This study, conducted by the USGS (Amjad et al., 1993) was performed in the upper Mojave River Basin in the high desert northeast of Los Angeles where 30 foot deep seepage pits were used to dispose of septage for a community of 46,000 residences. Depth to ground water is approximately 150 feet below ground surface. Using suction lysimeters to sample soil pore water in the unsaturated zone, this study found rapid nitrification of reduced organic compounds in the wastewater near the infiltration pits. However, nitrate concentrations decreased with depth, and nitrate concentrations in ground water samples from monitoring wells were less than 5 mg N/L. Two possible explanations for the low nitrate concentrations were suggested. First, the nitrate plume has not fully developed

due to the large depth to ground water. The vadose soils are primarily alluvial deposits and fan deposits ranging from unconsolidated to consolidated materials. A caliche layer underlies parts of the basin which results in low seepage rates locally. The vertical migration velocity of the wetting front reflects these diverse soil conditions and were estimated to range between 0.07 and 1.0 ft/d. This translates to travel times ranging from 6 years to 6 months. A second possible explanation of the low nitrate concentrations offered by the investigators is biological denitrification by autotrophic bacteria (chemolithotrophs). These are organisms which use reduced soil minerals, such as sulfides, as their energy source and nitrate as their electron donor. Little information was collected in this study to support this hypothesis. Also, Amjad et al. (1993) provide little data such as measurements of redox sensitive chemical species to assist in evaluating the redox conditions of the vadose zone or the underlying ground water.

In addition to nitrogen species, Amjad et al. (1993) analyzed selected anthropogenic organic compounds in their lysimeter samples but found no detectable priority pollutants at the four sites investigated. They also did not detect fecal coliform bacteria in any soil samples, even at samples collected just a few feet from the seepage pits.

1.6.   Summary of the Hydrogeology of the Albuquerque Basin

This section provides a general review of the ground water hydogeology and water quality in Bernalillo County, and surrounding areas. The purpose of this discussion is to establish a basic understanding of the occurrence, behavior, quality, and uses of ground water in the area. The Sandia and Monzano mountains serve as the boundary between the Albuquerque Basin and the East Mountaing study area. The geology of the two areas is dramatically different, accordingly, the ground water hydrology on the west and east side of the mountains will be discussed separately.

The seminal presentation of the geology of the Albuquerque Basin was prepared by Vincent Kelley and published in 1977 by the New Mexico Bureau of Mines and Mineral Resources (Kelley, 1977). More recent summaries of the geohydrology have been prepared by Hawley and Haase (1992) and Thorn et al. (1993). The Albuquerque Basin is part of the Rio Grande Rift (or depression) which is part of a rift valley system that is intermontaine through most of Colorado and New Mexico. The exact extent of the Basin is not clearly defined. Kelley (1977) attributes about 4,300 mi^2 to the basin, Thorn et al. (1993) ascribe a total area of 3,060 mi^2, and Hawley and Haase (1992) state that it covers and area of 2,100 mi^2. It ranges in width from 25 to 40 miles and is slightly over 100 miles long.

The Rio Grande Rift is described as a series of north-south trending basins which were formed by crustal extension initiated about 30 million years ago. Horizontal strain in the crust cause large blocks to sink and form basins between elevated mountain blocks. These basins subsequently filled with alluvial, lacustrine and eolian sediments which formed the high quality aquifers that are the major sources of water supply in Albuquerque and its vicinity. Although the Albuquerque Basin appears to be a single feature based on surface characteristics, recent studies cited by Hawley and Haase (1992) suggest it consists of two distinct structural basins (northern and southern). The northern half-graben dips to the east and a southern half-grabel dips to the west. These were formed by asymmetical groups of tilted fault blocks that were downdropped relative to adjacent mountain and plateau uplifts. Faults in the area represent important geological features that constitute many of the boundaries of the various geologic formations. Most are well mapped. A simplified map of the basin showing location of major boundary faults

is presented in Figure 0-3 and Figure 0-4 the associated cross sections are presented in (Hawley and Haase, 1992). A more complete mapping was published by Kelley (1977).

The boundaries of the Albuquerque Basin are generally associated with uplifts and related fault boundaries. The most significant feature is the eastern tilted Sandia-Manzano-Los Pinos uplift which forms the eastern boundary of the basin. These uplifted areas are composed of Precambrian plutonic and metamorphic rocks unconformably overlain by Paleozoic limestones and sandstones (Hawley and Haase, 1992). The Sandia Mountains are in the northern part of the Basin and Range physiographic province (Hawley, 1986) and cover about 400 square miles forming part of the eastern boundary of the Albuquerque Basin in central New Mexico. The highest point of the range is Sandia Crest at 10,678 ft. The Sandia Mountains are an eastward tilted fault block, gently dipping to the east with a steep west face. They are composed of a granitic pluton (spectacularly exposed on the west face) with a thin veneer of sedimentary rocks ranging in age from the Mississippian to Cretaceous (Kelley and Northrop, 1975).

The stratigraphy of the Albuquerque Basin can be subdivided into 3 units (Hawley and Haase, 1992): 1) pre-Santa Fe Tertiary deposits, 2) Santa Fe Group basin fill, and 3) post-Santa Fe Quaternary deposits (river valley and basin fill). The pre-Santa Fe Tertiary deposits underlie the Santa Fe Group, while the quaternary deposits are principally found along the river as a result of incision and backfilling events. The Santa Fe Group basin fill is the major fill unit of the Rio Grande rift basins and consists of alluvial material eroded from the adjacent mountains, with some localized eolian deposits. It ranges in age from 20 million years to 1 million years with the bulk of the deposition occurring between 5 and 15 million years ago as a result of fluvial processes. Santa Fe Group deposition ended about 1 million years ago when the Rio Grande and Rio Puerco started to cut their present valleys.

A major geologic feature that has special significance in this study is the location of the Sandia Fault (Figure 0-3). This fault extends in a north-south direction along the western toe of the Sandia Mountains approximately aligned with Tramway Boulevard. Its importance to this study is that this fault represents a vertical displacement between two geologic benches in the Upper Santa Fe formation of several hundred feet. The result is that to the east of this fault in the Sandia Heights subdivision the depth to ground water is less than 100 feet, while to the west of the fault in the North Albuquerque Acres area the depth to ground water can exceed 600 feet.
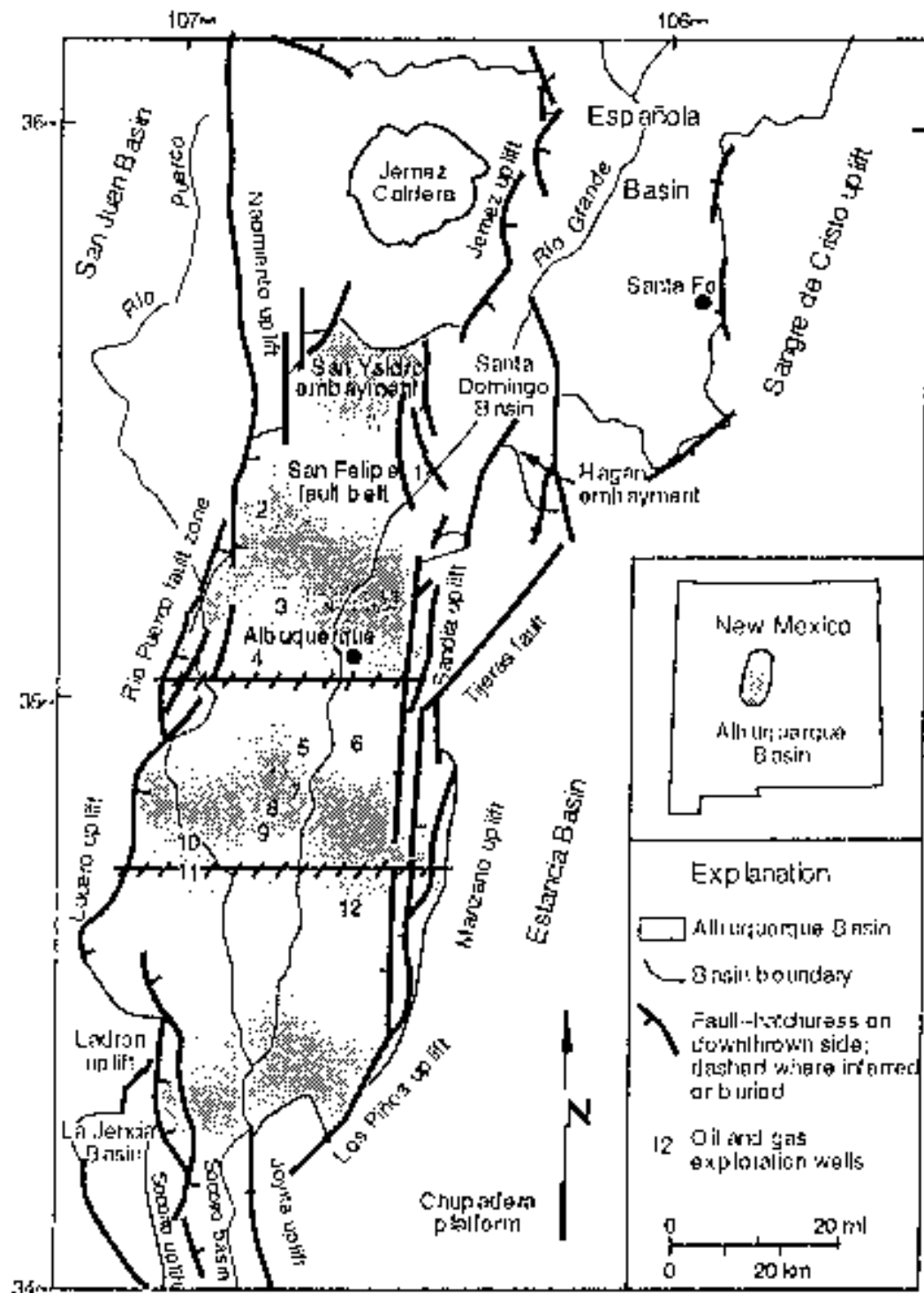
Figure 0-3. Index map of the Albuquerque Basin showing location of major boundary faults and geologic cross sections (Hawley & Haase, 1992).
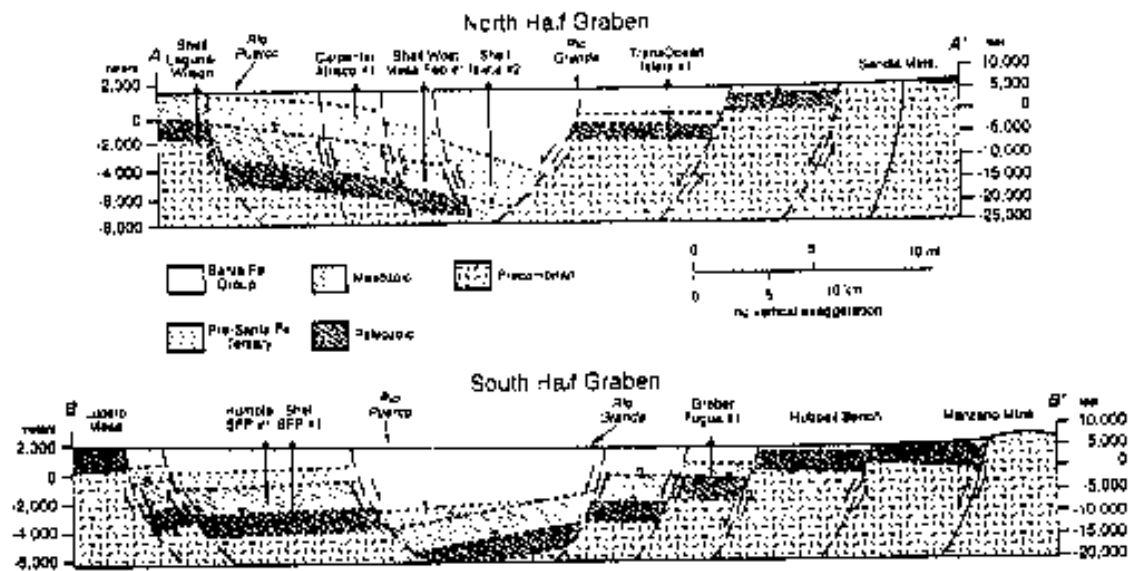
Figure 0-4. Diagrammatic geologic cross sections of central Albuquerque Basin (Hawley & Haase, 1992).

Hawley and Haase (1992) assembled the geologic information on the Albuquerque Basin for the purpose of developing a conceptual model of its ground water resources which was subsequently updated by Thorn et al. (1993). This information was then incorporated in a numerical model of the basin which has recently been published by Kernodle et al. (1994). This model is being used to assist in development and evaluation of management alternatives for meeting future water demand for the City of Albuquerque and the entire middle Riod Grande region, and has recently been approved for this use by the Office of the State Engineer (OSE).

1.7. Summary of Hydrogeology of the East Mountain Area

The geology and hydrogeology of the East Mountain area is much more complex than that on the west side of the Sandia and Manzano mountains. Important studies of this area include the pioneering work by Kelley and Northrop (1975), a comprehensive report by Titus (1980), and the more recent work by Turner (1990, 1992). A valuable feature of these reports is that they each contain a compilation of well records in the East Mountain area condensed from the NM State Engineer's files. This information constitutes the principal source of data which can be used to develop an interpretation and understanding of the occurrence and movement of ground water in the East Mountain area.

The Sandia and northern Manzano Mountains were formed as the result of uplifting and eastward tilting. The uplift exposed the Precambrian rocks, some dated at more than 1.3 billion years old, that give the Sandia Mountains their distinctive banded appearance (Titus, 1980). Prior to uplifting, these rocks were covered by at least 11,000 ft of sedimentary rocks of more recent geologic origin.

The Sandia and Manzano mountains are separated by the Tijeras and Gutierrez faults. These faults come together just west of the Village of Tijeras, and the region between the two is defined as the Tijeras basin (Titus, 1980). The location of important features and generalized geology of the East Mountain area are presented in Figure 0-5 (Kues, 1990). A cross section of the area is presented in Figure 0-6 (Kues, 1990).

Important descriptions of the geology of the East Mountain area have been provided by Kelley (1963), Kelley and Northrop (1975), and Bauer (1998). The main rock types in the East Mountain area were described by Titus (1980) based in large part on water well drilling records.

Furthermore, Titus (1980) characterized the ground water quality associated with each of these formations by analyzing and comparing their major ion chemistry (concentrations of Na, K, Ca, Mg, SO_4, Cl, and alkalinity). Titus' descriptions are briefly summarized in Table 0-3 generally in order of decreasing age.

Due to the complexity of the geology of the East Mountain area and the lack of properly constructed monitoring wells it is somewhat speculative to construct regional ground water potentiometric surfaces and associated flow patterns. Nevertheless, the reports by Titus (1980), Kues (1990), and Turner (1992) all contain plots of regional potentiometric elevations and/or ground water flow lines. The map developed by Kues (1990) is presented as Figure 0-7 Titus (1980) notes that as a first approximation, the contours of the potentiometric surface generally follow the form of the land surface, an observation that is particularly relevant to this study.

The area that was the focus of this study lies in a structurally complex area between the eastern toe of the Sandia Mountains and the Guitierrez fault (Figure 0-5 and Figure 0-6). The Tijeras fault lies at the boundary between the Sandia uplift and the Tijeras basin, and the Gutierrez fault bounds the Tijeras basin to the east. Within the Tijeras basin is the Tijeras anticline and Cedar Crest syncline (Kelley and Northrop, 1975). Kues
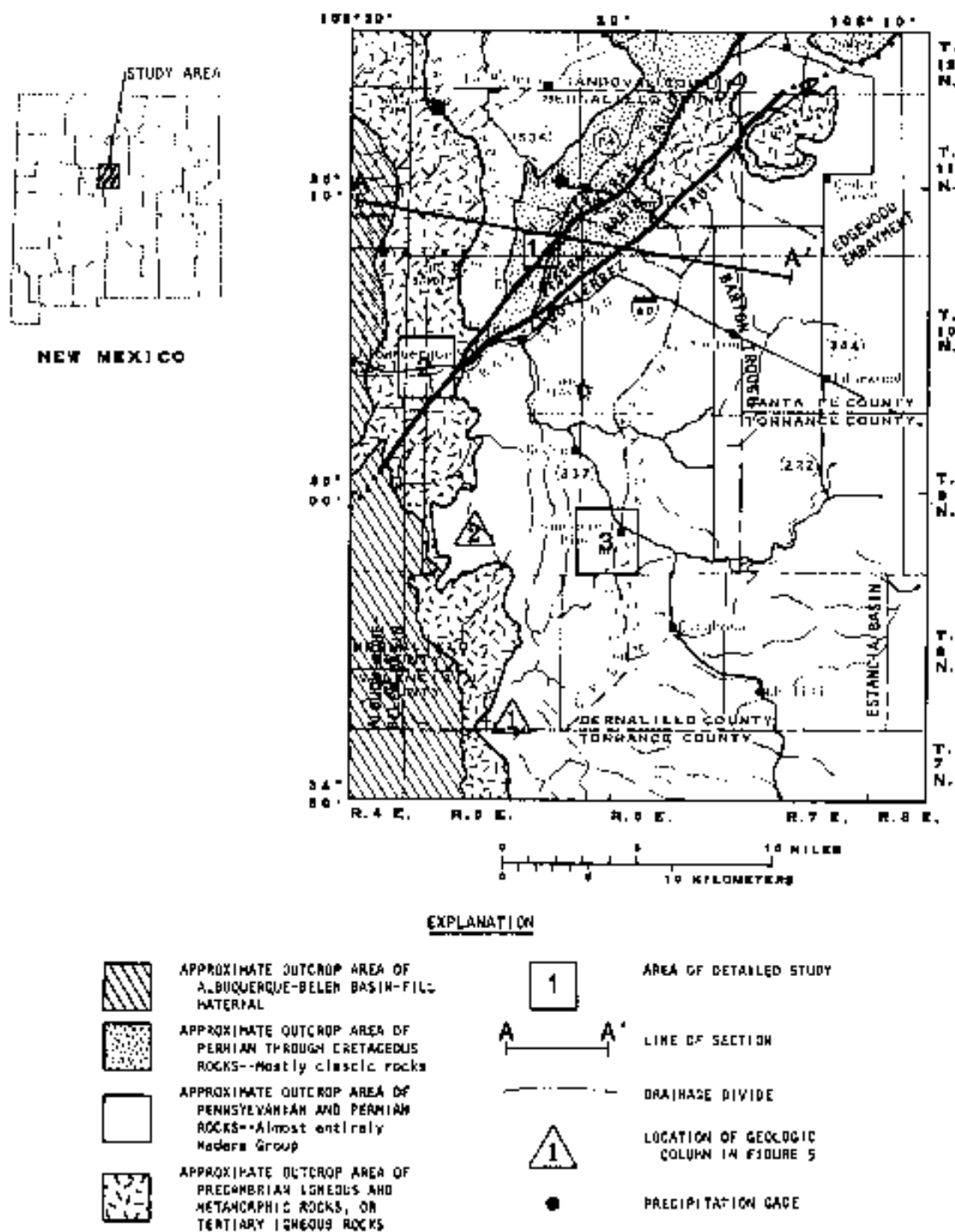
Figure 0-5. Index map of the East Mountain area showing location and generalized geology (Kues, 1990).
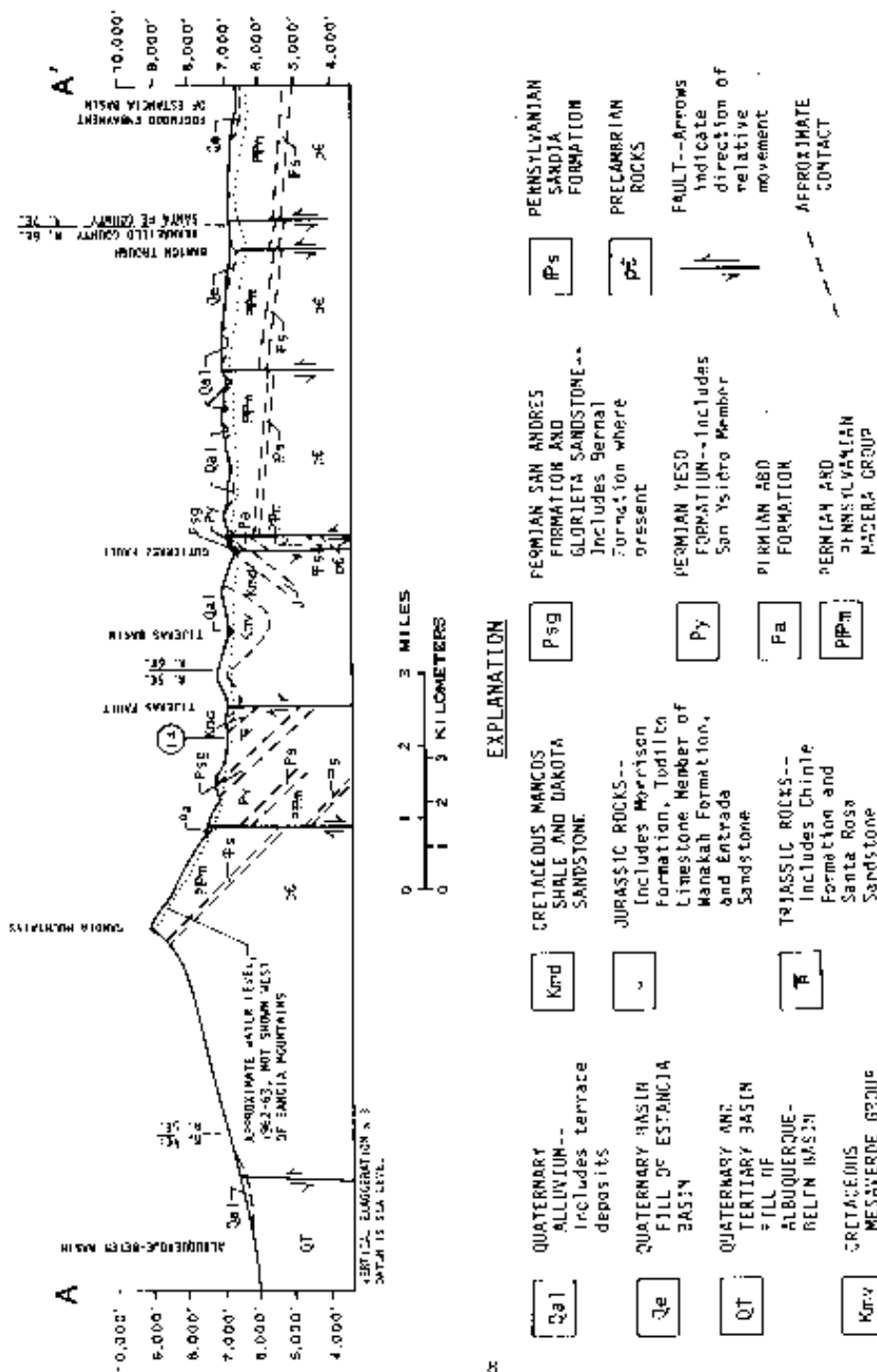
Figure 0-6. Geologic cross section A-A' of the East Mountain area (Kues, 1990).

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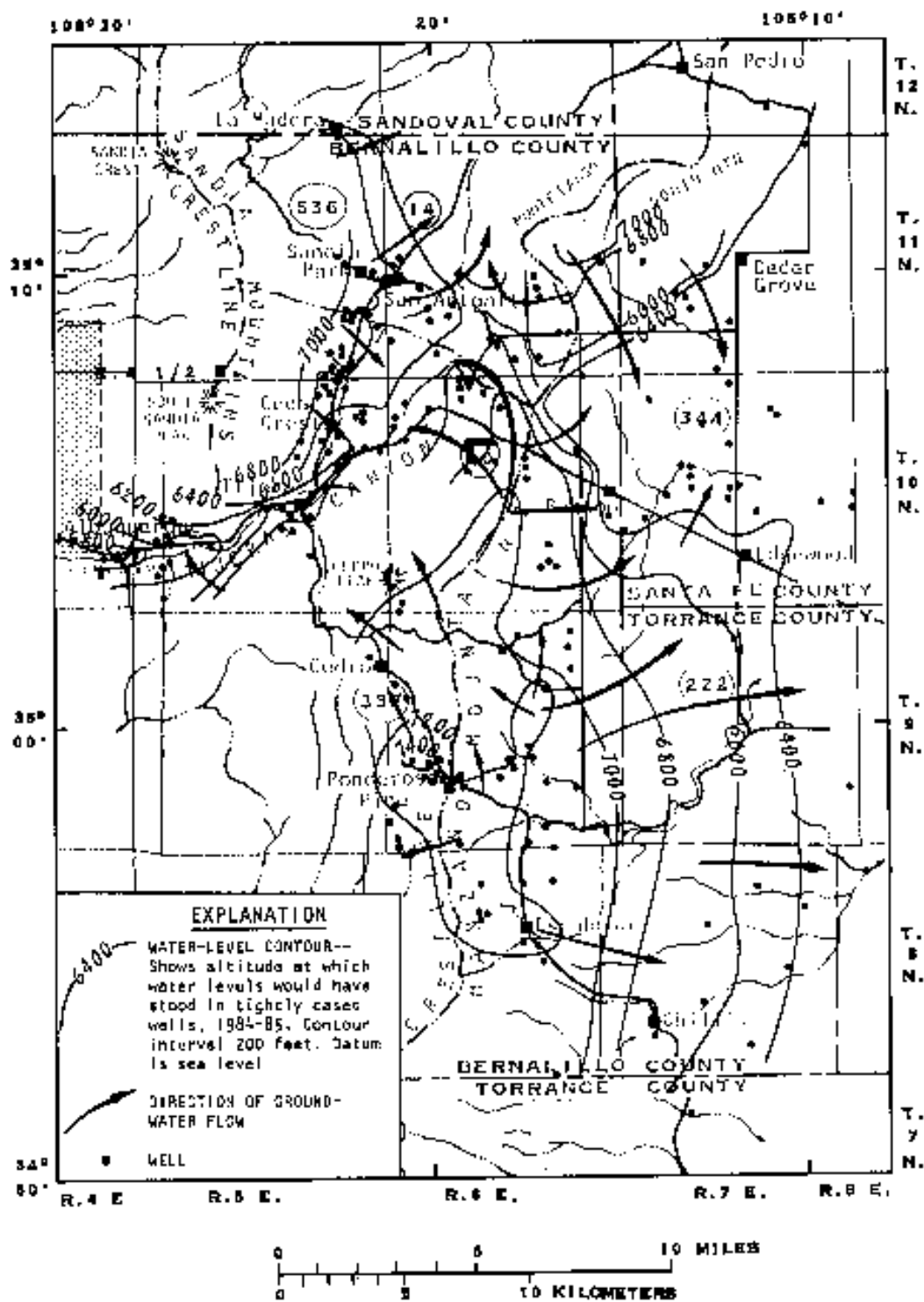
Figure 0-7. Ground water potentiometric surface of the East Mountain area (Kues, 1990).

Name	Rock Type	Ground water Availability
Precambrian	Metamorphosed sedimentary rocks, schists, greenstone, gneiss quartzite	Limited ground water resources
Sandia Formation	Interbedded shale, limestone, and sandstone	Limited ground water resources
Madera Limestone	Middle & Late Pennsylvanian limestone, highly fractured	Important ground water resource, but hard water
Abo Formation	Early Permian age sandstone beds	Probably poor aquifer
Yeso Formation	Early Permian sandstones, with some limestone	Good water quality, limited number of wells.
San Adres Limestone-Glorieta Sandstone	Early Permian limestone & sandstone	Adequate local water supply, high SO_4
Chinle Formation	Thick late Triassic mudstone with thin sandstone beds	Important aquifer for domestic wells, high SO_4
Santa Rosa Sandstone	Late Triassic sandstone & shale	Few wells, but adequate for domestic use
Morrison Formation	Late Jurassic multi-colored mudstone, sandstone, & some limestone	Water produced in fractures, very high SO_4
Mancos Shale-Dakota Sandstone	Thick Cretaceous marine shale, & sandstone	Poor water quality, high SO_4 & H_2S
Mesaverde Group	Late Cretaceous sandstones & shales	Poor quality water
Santa Fe Group	Miocene to Pleistocene alluvial fill	Important source of water in Albuquerque Basin
Estancia Valley Fill	Quaternary alluvial silts, sands & gravel	Important aquifer for domestic wells
Alluvium	Quaternary unconsolidated silts, sands & gravels along floors of mountain valleys	Important aquifer for domestic wells

Table 0-3. Major rock units of the East Mountain area (from Titus, 1980).

The Pennsylvanian Madera Group, consisting of marine limestones and shales form the predominate aquifer in the region (Titus, 1980). However, within the study area Permian through Cretaceous sedimentary formations are the principal aquifers, cropping out within the study area and dipping generally to the east 30º - 35º. The water table in the study area generally follows

the topography, but is also controlled by the geologic structure in the area. The Tijeras Fault bisects the area northeast to southwest and causes the shallow groundwater in contact with it to flow southward into Tijeras Canyon and into the Rio Grande Basin. There is an east-west local divide that approximately coincides with the Frost Arroyo that splits the north and south outflow from the area. Water to the south of the divide flows out Tijeras Canyon, and water to the north of the divide flows north along San Pedro Creek. The three subdivisions included in this study are all south of this divide.

In the Sierra Vista subdivision the Tijeras Anticline forms a topographic high and creates a local groundwater divide. The water table is located in the underlying Cretaceous Mancos Formation, a marine shale, and is undesirable for domestic purposes due to high salinity and the formation generally doesn't produce water in usable quantities (< 0.25 gpm). The water has a total dissolved solids (TDS) concentration of 1180 mg/L and a slight odor of sulfide-gas. Field measurements of dissolved oxygen show very low levels (0.5 - 1.5 mg/L) and analyses give low concentrations of sulfates. These indicate that reducing conditions exist at least locally in and near the well. This may be a perched aquifer within the topographic high.

The water table in Piñon Ridge is also in the Mancos Formation, but of better quality and supply (Titus, 1980). This saturated zone is weakly connected to the underlying aquifer in the Dakota Formation (marine sandstone, Kelley and Northrop, 1975). The Dakota is the main source for the domestic wells in the subdivision with adequate supply and quality. Water level measurements in two domestic wells (screened in the Dakota) less than one hundred feet from the monitoring well show that the Mancos is at least partially confining the Dakota. The monitoring well water level is ~ 170 feet below the surface, the domestic wells have water level measurements at ~ 320 feet below the surface. Well record logs indicate that the contact between the Mancos and the Dakota is approximately 280 feet beneath the surface.

Sandia Park is underlain by eastward tilted sedimentary beds ranging from Permian to Triassic in age. The water table cuts horizontally across the contacts generally dipping to the east to follow the topography of the area. The main aquifers of the area are the Permian San Andres Formation, a marine sandstone and limestone, and the Triassic Santa Rosa and Chinle Formations, fluvial and flood-plain sandstones and mudstones. The water quality is good for domestic use from both formations and most wells produce water in usable quantities (Titus,

1980). The presence of confined layers at shallow depths (i.e. less than 100 ft) was not reported by Titus (1980). This is important because a horizontal zone of low conductivity might delay contaminants from on-site disposal systems from reaching ground water.

1.8.   Previous Studies of Ground Water Contamination

A general review of ground water contamination in the Albuquerque Basin and in Bernalillo County was prepared by CH2M-Hill (1990). This report was based on records from the NM Environmental Improvement Division (predecessor to the NMED), the Albuquerque Environmental Health Department, the Bernalillo County Environmental Health Department, the NM Oil Conservation Division, the Bureau of Indian Affairs, the US Department of Defense, the US Department of Energy, and the Southwest Research and Information Center. This data was entered into a geographic information system so that maps of contamination could be generated. However, little data on regional ground water contamination from on-site wastewater disposal systems was available for most of Bernalillo County.

Two notable studies of ground water contamination in the inner valley of the Rio Grande have been conducted. Heggen et al. (1980) investigated ground water contamination in the North Valley of Albuquerque, and Gallaher et al. (1987) investigated ground water contamination in the South Valley. Both of these studies found little nitrate contamination in regions where shallow ground water resulted in insufficient reaction time and limited O_2 transfer so that the nitrification process (reaction (2.9)) does not occur. Furthermore, evidence of reducing conditions was noted throughout the inner valley which includes high concentrations of Fe(II), Mn(II), and H_2S in ground water samples. Outside of the inner valley where depth to ground water exceeds about 20 ft, nitrification of leachate from on-site disposal systems occurs and nitrate contamination of underlying ground water resources is found. It should be noted that lack of nitrifying conditions which therefore prevents nitrate contamination of underlying ground water is not appropriate justification to permit high densities of conventional on-site wastewater systems as other ground water quality problems may result.

There have been few investigations of ground water contamination from on-site wastewater disposal systems outside of the inner valley of the Rio Grande in Bernalillo County. The most comprehensive data has been collected by the USGS in three fairly recent monitoring programs which relied on water samples obtained from domestic water wells (Kues, 1990; Kues

and Garcia, 1995; Rankin, 1995). Kues (1990) analyzed water samples from the East Mountain area for NO_2^- and NO_3^- and found concentrations exceeding 5 mg N/L in 8 of 26 wells. The highest nitrate measured concentration were found in the Carnuel area of Tijeras Canyon. Elevated concentrations were also found in the Sandia Park area and in a well in Chilili located in far southeastern Bernalillo County.

An investigation by Kues and Garcia (1995) consisted of repeated measurements of ground water quality and ground water levels from 81 wells in 4 unincorporated areas of Bernalillo County from 1990 to 1993. These consisted of 20 wells in the East Mountain area, 11 wells in North Albuquerque Acres and Sandia Heights, 20 wells in the Rio Grande Valley immediately north of Albuquerque, and 30 wells in the South Valley area. Of most relevance to this study were the wells in the East Mountain area and the North Albuquerque Acres/Sandia Heights subdivisions (the locations of the East Mountain area wells are presented in Figure 0-8). The wells in the North Albuquerque Acres/Sandia Heights area were all over 360 feet deep, and most were greater than 500 feet deep. No ground water samples collected from this area had NO_3^- greater than 5.0 mg/L as N, however, 3 of the 11 wells did have combined concentrations of NO_2^- and NO_3^- of greater than 2.5 mg/L as N, suggesting N concentrations slightly elevated above background values.

In contrast to the North Albuquerque Acreas/Sandia Heights area, all of the water wells in the East Mountain except one were less than 300 feet deep, and 10 of 16 with known well elevations were less than 200 feet deep. Six of the 20 domestic water wells sampled in the East Mountain area had NO_3^- concentrations that were consistently greater than 5 mg/L as N, and at least 5 others had definite trends showing increasing nitrate concentration. The major areas of nitrate contamination were in Tijeras Canyon, along I-40 (near Zuzax and Barton), and Sandia Park.
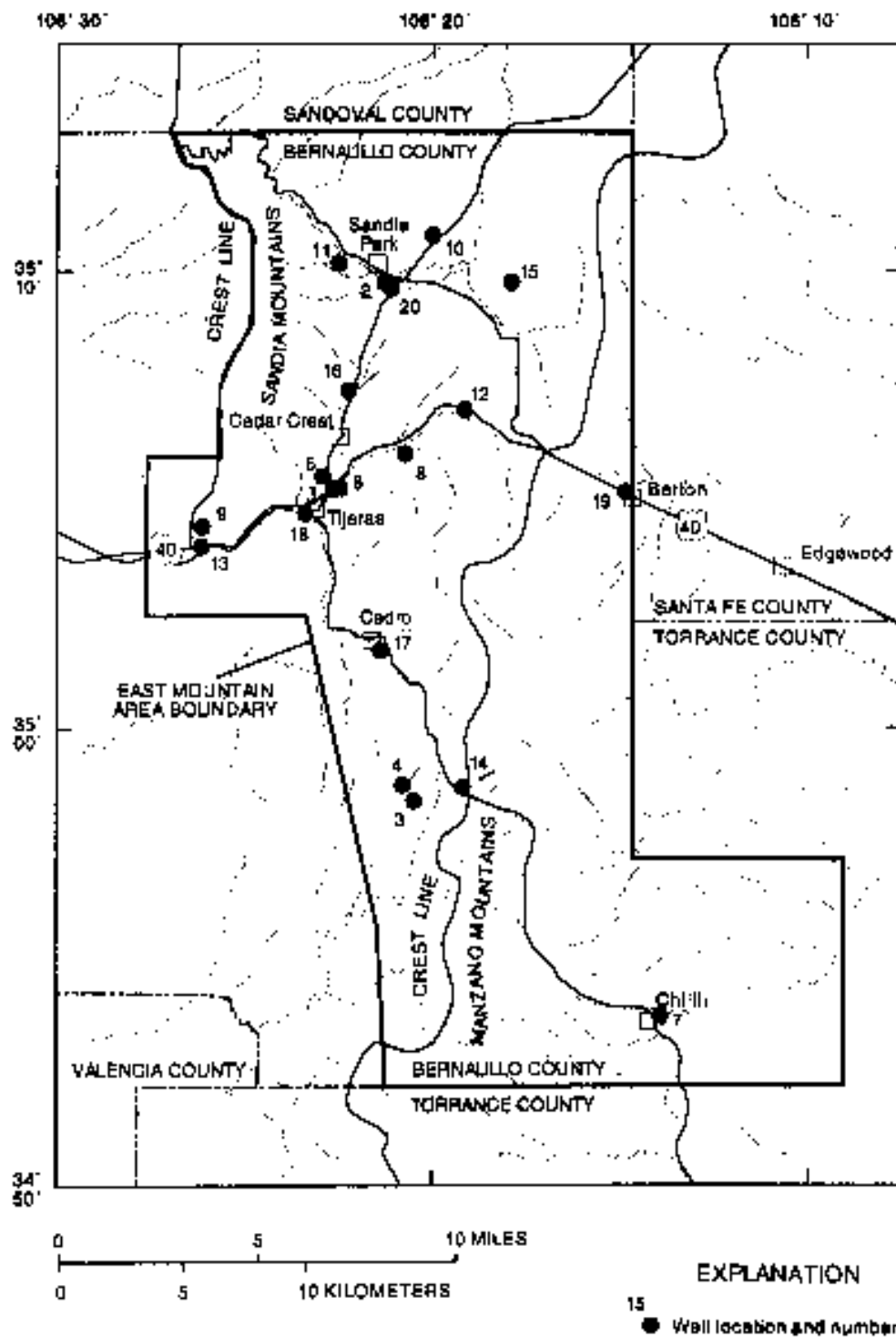
Figure 0-8. Location of East Mountain area wells sampled by Kues and Garcia (1995). Large diameter dots represent wells with nitrate concentrations greater than 10 mg N/L.

Rankin collected water samples from 75 wells in unsewered parts of Bernalillo County during the summer of 1995. Fifteen of the wells had nitrate concentrations greater than 5 mg N/L. Wells sampled in the Sandia Park, Cañoncito, Piñon Ridge, and Sierra Vista all had elevated nitrate concentrations. Of particular relevance to this study is a domestic well near the intersection of NM Highways 14 and 536 (the Crest Highway) which routinely produces water samples containing 20 mg N/L (well number 20 in Figure 0-8). This well is approximately 200 m (650 ft) south of monitoring well SP01 constructed in this study. No construction details on this well are available.

In reviewing the information on ground water collected to date it is important to remember that virtually all of the ground water samples collected in unsewered areas of Bernalillo County to date have come from domestic water supply wells. A major confounding factor in analyzing water quality data from domestic wells is that there is often little reliable information on how the well was constructed including its depth, the location of the well screen, or whether the annular space between the casing and the hole was sealed to prevent vertical flow of water to/from overlying water bearing strata. Furthermore, domestic wells are typically constructed to collect water from well below the top of the water table, whereas contamination from on-site disposal systems will initially occur at the top of the water table. Except in unusual circumstances, vertical migration of contaminants through ground water is a slow process, principally controlled by diffusion. Therefore, evidence of contamination in water samples from domestic wells may not occur until long after the discharge of pollutants by which time the aquifer may be severely contaminated. Considering these uncertainties, as well as the fact that 20% of the most recent wells sampled by the USGS had nitrate concentrations greater than 5 mg N/L, the conclusion may be reached that there is significant ground water contamination by on-site wastewater disposal systems in unsewered areas of the County.

DESCRIPTION OF THE GROUND WATER AND VADOSE ZONE INVESTIGATIONS

1.9.   Ground Water Investigation

There are three particular areas of concern in Bernalillo County where regional ground water contamination is likely to occur as a result of on-site wastewater disposal systems: the inner valley of the Rio Grande and adjacent areas, rapidly developing unsewered subdivisions on the western slope of the Sandia Mountains including Sandia Heights and North Albuquerque Acres, and unsewered communities and subdivisions in the East Mountain area both north and south of Interstate 40. Due to the high density of development within the inner valley, the City of Albuquerque and Bernalillo County have made a joint commitment to provide community water and sewer services throughout the inner valley of the Rio Grande using a combination of federal, state, and local funding sources. The intention is to solve the ground water quality problems by collecting and treating all domestic wastewater in this area. Therefore, the study described in this report has focused on the unsewered areas of Sandia Heights/North Albuquerque Acres and the East Mountain Area, where providing public water and sewer service is not planned.

To a large extent, this exercise is analogous to prospecting for a valuable mineral such as gold or oil. It is widely accepted that ground water contamination has occurred from on-site disposal systems. One of the principal objectives of this study was to test this hypothesis. Therefore, careful consideration was given to locating study areas which had conditions believed to be most conducive to detection of ground water contamination from on-site systems.

1.9.1.   Criteria for Selecting Study Areas

A major objective of this investigation was to identify ground water contamination that has occurred as a result of on-site wastewater treatment and disposal systems. Ideally this would be accomplished by locating unsewered communities or subdivisions underlain by a described and characterized water table aquifer. A series of up gradient and down gradient monitoring wells would be installed and the deterioration in water quality as ground water passed beneath the community would be attributable to the on-site disposal systems. This scenario was

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reasonably representative of conditions in the Sandia Heights development on the west side of the Sandia Mountains. However, the geology of the East Mountain area is so complex and heterogeneous that the ability to identify a water table aquifer in many areas is highly problematic. This further complicates the difficulties associated with quantifying ground water contamination resulting from on-site systems.

A set of criteria was developed to assist in identifying areas with the greatest likelihood of being able to document regional ground water contamination. These are summarized in Table 0-1.

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1. Conditions in the area suggest a likelihood of finding ground water that has been contaminated by on-site wastewater disposal systems:
   a. Hydrogeologic data is available which suggests the presence of a water table aquifer
   b. Ground water quality data is available from previous studies
   c. The contaminated aquifer is of sufficient areal extent that multiple wells could be completed to delineate the extent of contamination
   d. Depth to ground water is not excessive.
2. A community is present with a relatively high density of on-site wastewater disposal systems.
3. Accessible drilling sites are available.
4. Existing monitoring wells are present or nearby.

Table 0-1. Criteria developed to assist in identifying areas for further study.

The first criteria listed in Table 0-1 calls for identification of an area where the ground water hydrology is well understood as a result of previous hydrogeologic investigations. Ideally data would be available regarding well records, stratigraphy, location and extent of underlying ground water formations, and hydraulic properties of the aquifer. In addition ground water quality data from previous sampling programs (i.e. the USGS ground water monitoring programs) would provide some indication of potential water quality problems.

The second criteria can be evaluated, for the most part, using data available from County subdivision plats, aerial photos, and building permits. A series of air photos taken in 1993

provided by the County was found to be especially useful because they provide information on the density of development of an area, and also provide some insight on the topography and geological features.

The third criteria addresses accessibility. In developing these selection criteria it was assumed that construction of ground water monitoring wells on private property would be undesirable because ground water contamination may be viewed by the land owner as compromising the land's future property value. Other concerns with locating monitoring wells on private property would be assuring access to the wells in the future, and the potential problems associated with operating large drilling equipment in the vicinity of a residence. Therefore, one of the selection criteria was the ability to locate monitoring wells on public property, including road easements or dedicated public land (e.g. open space, State or Federal property, or public school land).

To date no studies of ground water quality in either the Sandia Heights/North Albuquerque Acres or East Mountain areas has involved collection of samples from monitoring wells. However, there are several monitoring wells in the area which have been constructed as a result of soil and ground water contamination at LUST sites. Information provided by the Underground Storage Tank Bureau of the NMED suggest that there are at least four LUST sites in the East Mountain area at which ground water monitoring wells have been constructed (Crandall, personal communication, 1995). These sites are: (1) the Tijeras Fire State East Command Center in Tijeras; (2) Lee's Gung Fu in Cedar Crest; (3) the Indian Hills, Snodgrass Well, and Canyon Auto sites all located along I-40 and east of Tijeras; and (4) the Caristos site west of Edgewood. While these wells have been monitored extensively for petroleum hydrocarbon compounds as part of the investigation and remediation process, water quality parameters indicative of contamination from on-site disposal systems have not been previously analyzed (i.e. nitrate, coliform bacteria, etc.). (Recent acceptance of natural attenuation as a remediation strategy at LUST sites has caused the NMED to request analysis of some ground water samples for indications of microbial growth, which usually includes some of these parameters.)

Based on the information collected at the start of this study, it was concluded that Sandia Heights and four communities in the East Mountain area (Sandia Park, Piñon Ridge Estates,

Sierra Vista Estates, and Sandia Knolls) satisfied some or all of the criteria identified in Table 0-1. Other areas which were considered as possible sites for ground water investigations included: Tablazon, Zuzax, Cedar Crest, Cañoncito, and Cedro. These were rejected principally due to the lack of information regarding hydrogeological interpretation, uncertainty about the presence of a water table aquifer, and low residential housing density. The Tablazon community was not considered because it is served by a community wastewater collection and treatment system. The characteristics of each community are summarized in Table 0-2.

Name	Date First Platted	Approx. Density	Depth to Ground water	No. of Wells	Water Quality Data Available?
First Priority					
Sandia Heights	1928	1/acre	200-400 ft	Few	3 USGS wells
Piñon Ridge	1960	2/acre	200-300 ft	Many	1 USGS well
Sandia Park	1927	1/acre	50-200 ft	Many	2 USGS wells
Sierra Vista	1961	2/acre	350 ft ?	Few	No
Sandia Knolls	1956	1/acre	50-250 ft	Many	1 USGS well
Zuzax					LUST Site
Cedar Crest	1980's		20-150 ft	Many	LUST Site
Second Priority					
Cañoncito			50-200	Many	1 USGS well
Tablazon	1956	1/acre			No
Cedro	1970's		30 ft	Some	1 USGS well

Table 0-2. Summary of characteristics of candidate communities or subdivisions for ground water monitoring.

The communities in Table 0-2 are listed in two priorities. The highest priority areas were assigned to communities or subdivisions which have water table aquifers that are reasonably well delineated and contain high densities of residences served by on-site disposal systems. In addition, high priorities were assigned to locations with nearby LUST sites as hydrogeologic data and water quality information could be collected quickly and at relatively low cost. The field study focused on the high priority locations listed in Table 3-2.

The Sandia Heights neighborhood lies at the base of the western face of the Sandia Mountains. The geology of this area is summarized in Section 2.2. The Sandia Heights study area, at an altitude of 6,200 ft., overlies the piedmont-slope alluvium, a coarse-grained deposit of

mostly granitic detritus forming coalescent fans off the uplifted face of the Sandias. This alluvium overlies the Upper Santa Fe 1 (USF-1) unit, which consists of older Pleistocene coarse-grained alluvial deposits, a sub-unit of the Upper Santa Fe (USF) (Hawley and Haase, 1992). A north-trending, down to the west, fault zone (< 1 mile wide) under the center of the study area drops the Sandia granite and the water table, several hundred feet (Connell, 1998).

The mean annual precipitation in Sandia Heights ranges from 12 - 14 inches per year (Thorn, et. al., 1993), resulting in a semi-arid climate. Surface water in Sandia Heights is present only in ephemeral arroyo beds during storm runoff. All water supply is from the unconfined aquifer that is recharged by mountain front runoff from the west face of the Sandia Mountains. The water table to the east of Sandia Heights is perched on top of the buried granite and is comparatively shallow (28 feet below the surface from the SEO well record for the Ellena Gallegos well) compared to the water table west of the neighborhood (> 450 feet). The main control on the depth of the water table is the north-trending, down to the west Sandia Faults. The water level drops more than 400 feet across the pair of faults. Ground water flow is generally to the west toward the river and has an average gradient of 0.11 across the area. Data parameters for the aquifer and study area and the references are given in Table 0-3. Figure 0-1 shows the topography, mapped locations of the faults in the study area, and the water table elevations in the monitoring wells.
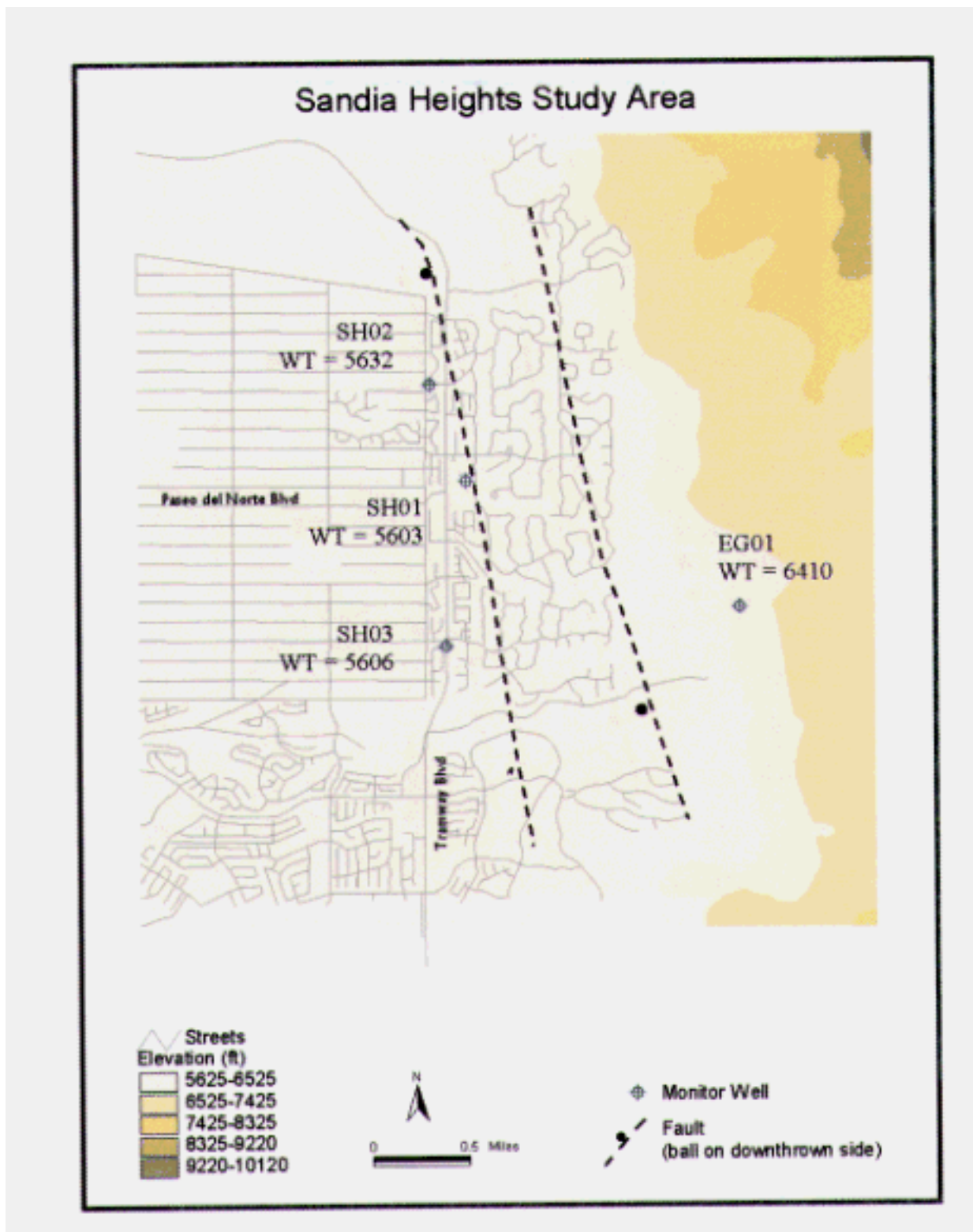
Figure 0-1. Map of the Sandia Heights Study area.

Annual Rainfall	12 - 14 in	NMBMMR OFR 425 Connell '96 p. 28
Annual Recharge	12.21 cfs	USGS Report 93-4149 fig 39 p. 86 (Hawley&Haase)
Depth to Water	<100 - 500 ft	SEO Well records
Soil Type	Moderately developed argillic and calcic soil	Connell, '96 p. 95 vol 1
Soil Depth	at least 15m	Connell, '96 p. 95 vol 1
Depth to Bedrock	<100 to > 3000 ft	Hawley & Haase '92 Plate 2
Aquifer Type	Unconfined	
Aquifer Thickness	<50 feet (perched on the granite)　to > 450 feet	
Aquifer Composition	Gravelly sand, silt and clay; lenses of sand, gravel, and silty clay (USF-1, PA)	Hawley & Haase '92 Table III-2, p. III-11
Area of Subdivision	854 acres	Albuquerque GIS data base
Average Lot Size	.8 acre	Albuquerque GIS data base
Number of Residences	1046	Albuquerque GIS data base
Number of Septic Tanks	1046	Albuquerque GIS data base
Specific Yield	.15	USGS Report 94-4251 p. 110
Hydraulic Conductivity	4 ft/day	USGS (McAda) MRGB Workshop 11 Feb 98

Table 0-3. Data parameters and references for the aquifer and study area.

The East Mountain area that was the subject of this study is located entirely within the Sandia Basin as designated by the New Mexico Office of the State Engineer. This basin lies at the base of the eastern slope of the Sandia Mountains, and it is interesting to note that its boundaries do not coincide with hydrologic boundaries (

Figure 0-2). A preliminary summary of the hydrology of this basin has recently been published by Peterson (1999). To the west, the basin is hydrologically bounded by the crest of the Sandia Mountains, and to the east by the groundwater divide between the Rio Grande basin and the Estancia basin. Flow out of the basin is to the north and south. Northerly flow is along San Pedro Creek turning west at the north end of the Sandias and westward into the Rio Grande basin. The southerly flow is into Tijeras Canyon and westward again, to the Rio Grande basin (Titus, 1980). Recharge to the area is dominated by the precipitation that falls on the east slopes of the Sandia Mountains (Kues, 1990). Average annual precipitation for Sandia Park is 18.9 inches per year and for Sandia Crest is 22.9 inches. Regional recharge rates for this basin have been estimated to range between 0.5 and 1.0 in/yr (Peterson, 1999). A diagram of the Sandia Basin is presented in

Figure 0-2).

Water table elevation data was a key factor in choosing drilling sites for the East Mountain monitoring wells. The water table needs to be accurately identified for three reasons. First, locating ground water in many parts of eastern Bernalillo County is often highly uncertain due to the heterogeneity of the geologic formations. Second, a true water table aquifer is desirable from a monitoring perspective as an unconfined water bearing strata is more likely to be contaminated by on-site disposal systems than groundwater protected by an overlying impermeable confining layer. Knowledge of the depth of the water table is needed so that short lengths of well screen can be placed across the top of the saturated zone to capture water most directly influenced by on-site disposal systems. Finally, a better estimate of the depth of water leads to better choices of drilling sites.
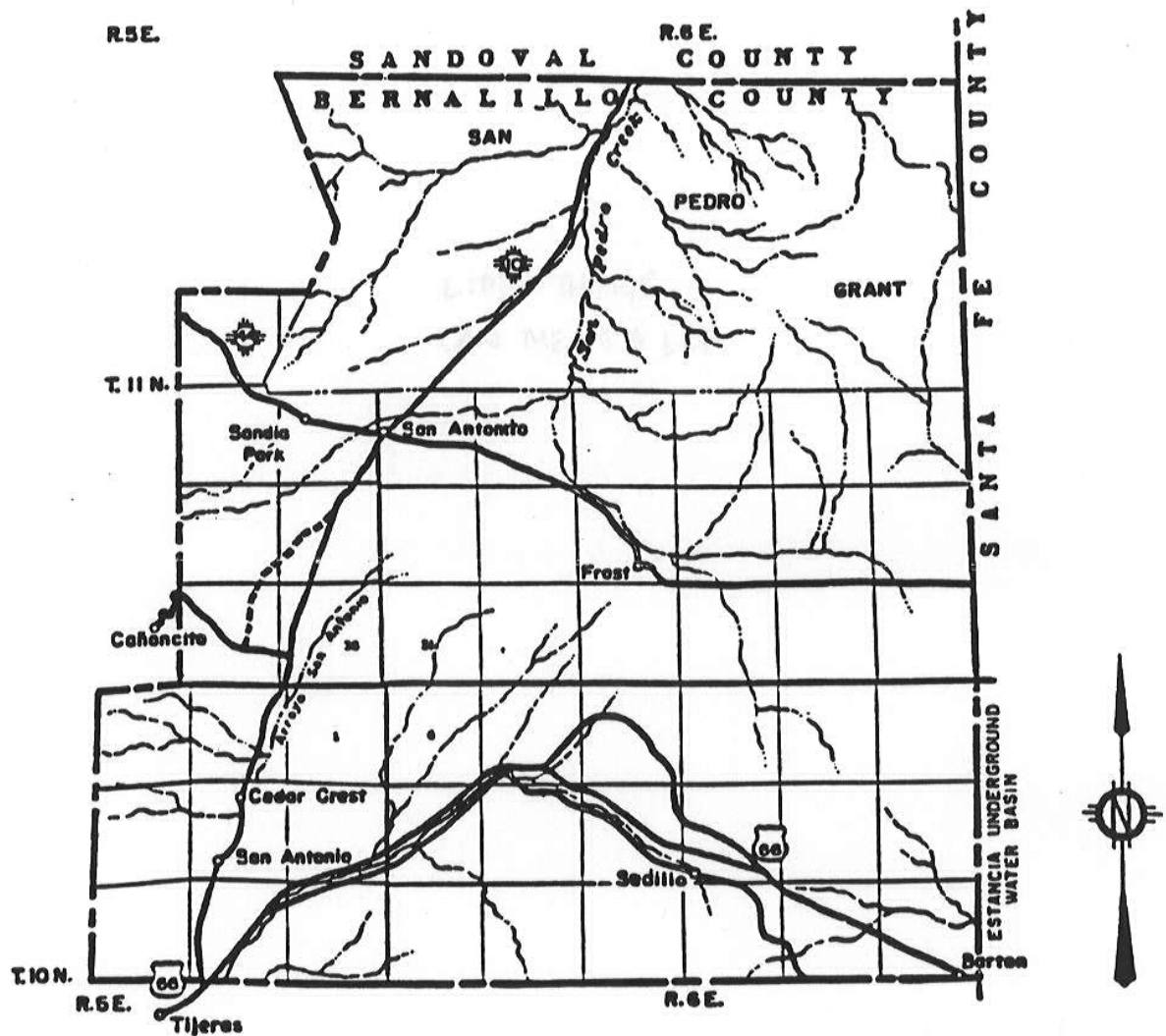
Figure 0-2. Map of the Sandia Basin (Office of the State Engineer).

Water table elevation maps of this region have been done by Titus (1980) and Kues (1990) using available water level data, however in both cases the area mapped is so broad that only regional trends in ground water elevations could be shown. Localized waste level data was needed in the vicinity of the subdivisions considered in this investigation, therefore a new map of water table elevations specific to the three study areas investigated in this project was

constructed. Accordingly, a ground water elevation map was prepared of the four study areas considered to assist in selecting locations for monitoring wells.

The water table elevation map was drawn on a digital elevation map of the East Mountain area using ground water elevation data compiled by the Office of the State Engineer. Prior to constructing this map, a map showing the local geologic features was developed using information from Kelley (1977), Titus (1980), and Kues (1990). Next, the domestic water well records filed by drillers and maintained by the Office of the State Engineer were examined. Most of these files described wells that were located by lot number and subdivision, hence the first plotting of the wells was done on zone atlas maps. Records were chosen on the basis of date (most recent records have the most current water table elevation data), completeness of location data, quality of well logs, and completeness and quality of water level and casing data. Approximately 250 records were obtained for the Sandia Park, Sierra Vista, and Piñon Ridge subdivisions in the East Mountain area. The well locations were plotted on each zone map with the depth to water, site elevation interpreted from USGS 7.5 minute topographic maps, and water elevation above sea level (site elevation minus depth to water). Next, an overlay of points with only the water elevation was made from the zone atlas maps. After adjusting for different scales, this was overlaid onto a topographic map in order to construct water level contours. The contouring was done as an overlay on the topographic base map because the water table generally follows the terrain, so the water elevation and the terrain are both used to make the contours. The final water table elevation map was drawn by an iterative process in which the location of contours was determined by interpolation between nearby wells, contours plotted, and then adjusted to produce a map that did not contain physically impossible flow regimes or unreasonable singularity points. It is important to recognize that any regional map of ground water elevations requires use of hydrologic judgement associated with interpreting uncertain field data and complexities due to unusual flow regimes. The map prepared for this study is no different, with the principal uncertainties being the result of 1) uncertain surface elevations of the well location, 2) incorporation of data ranging over several years in age, 3) uncertainties regarding the geologic strata reported in the drillers' logs, and 4) unusual ground water hydrology due to the prevalence of fracture flow patterns in the study area.

Along with the water table data, subsurface geometry of the hydrologic units and the geologic structures is needed. Geologic cross-sections were made near Sandia Heights, Piñon Ridge, and Sierra Vista from diagrams constructed by Kelley and Northrop (1975). The anticline (upward folding) structure under the Sierra Vista subdivision and the fault through Sandia Heights that drops the Precambrian granite (and so the water table) are examples of the type of significant data that these cross-sections show.   Figure 0-3 is a map summarizing the geologic structure of the East Mountain study area and Figure 0-4 is a contour plot of the water table elevations in this area.
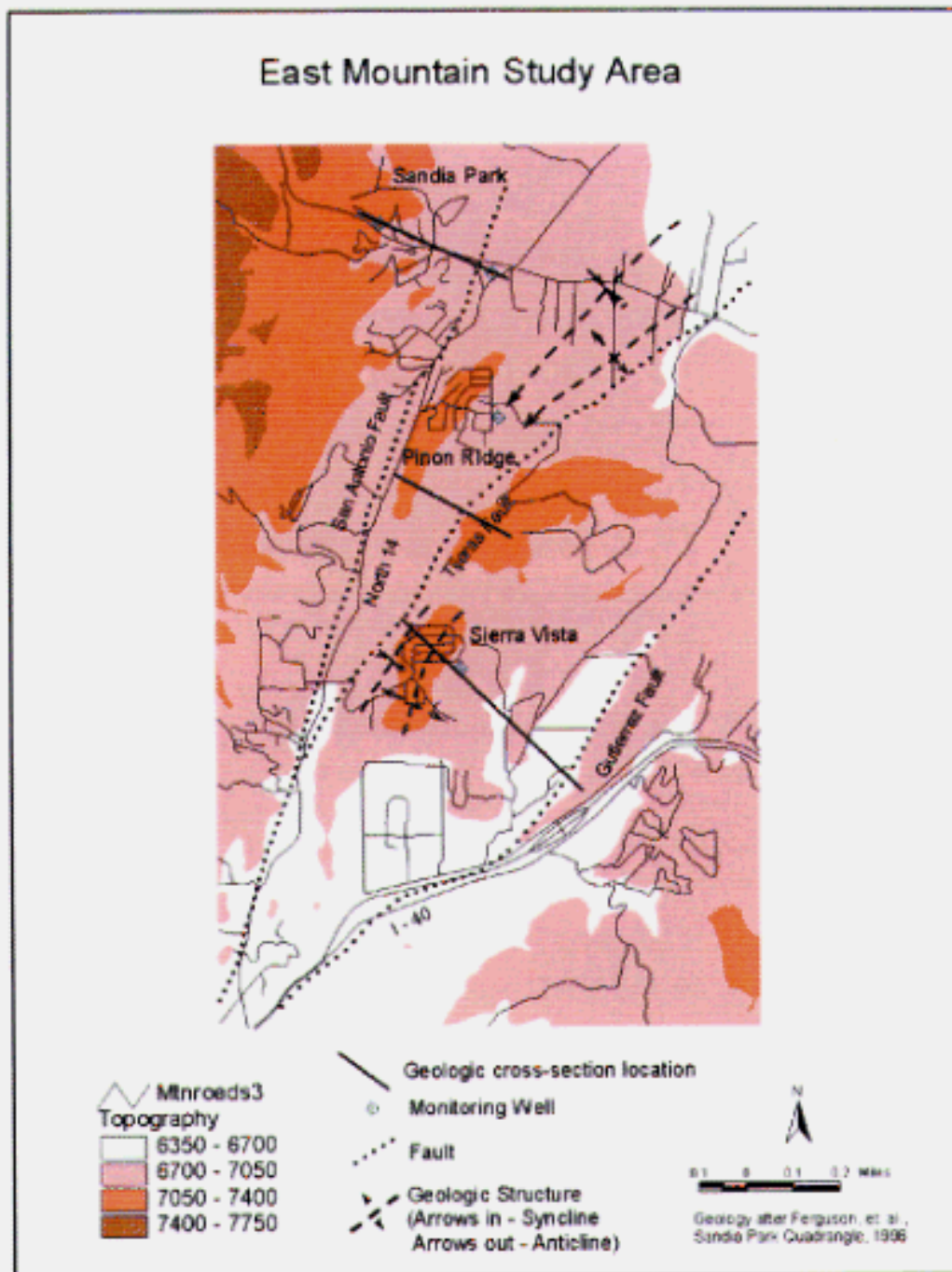
Figure 0-3. Principal geologic structures in the East Mountain study area.
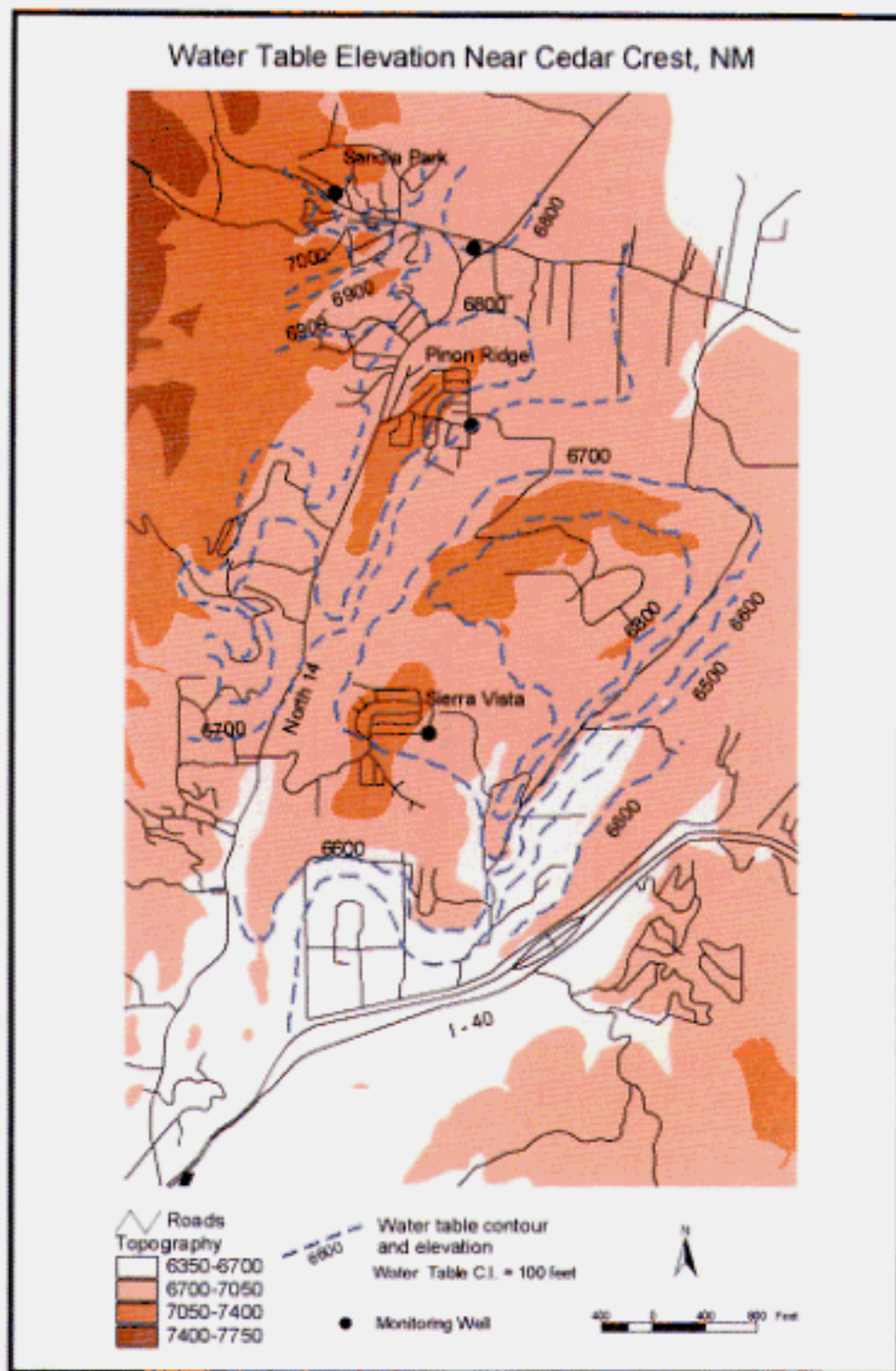
Figure 0-4. Contour map of water table elevations in the East Mountain study area. Monitor well locations indicated by •.

1.9.2. Ground Water Sampling from Existing Monitoring Wells

Two LUST sites were identified for sampling in the East Mountain area; Lee's Gung Fu and Indian Hills. The Lee's Gung Fu LUST site is located at the Texaco gas station in Cedar Crest, NM along NM State Highway 14 just west of the Sierra Vista subdivision. The Indian Hills LUST site is located at the Interstate 40 Zuzax exit 175 in Tijeras Canyon. Two monitoring wells were sampled at each site in March of 1997. The Lee's Gung Fu monitoring wells are completed in a thin shallow alluvial aquifer on top of the relatively impermeable Mancos Shale with depth to water of approximately 20 feet. The Indian Hills wells are completed in highly faulted Permian/Pennsylvanian Formations with an average depth to ground water of 50 feet. Monitoring wells T8 and F10 were sampled at Lee's, and water samples were collected from MW 01 and MW 10 at Indian Hills. The results are given below in Table 0-4. Elevated nitrates were found at Lee's Gung Fu in both of the wells tested, and in one of the Indian Hills wells. However, these sites are not located in close proximity to any developments with high densities of on-site wastewater disposal systems and so were not considered for installation of additional monitoring wells.

Wells	Sample Date	Chlorides (mg/L)	MBAS (mg/L)	Nitrates (mg/L)	E. Coli. 1= present 0 = absent	Total Coli. 1= present 0 = absent
Lee's Gung Fu						
T8	3/4/97	47.3	0	2.69	0	1
F10	3/4/97	135	0	3.39	0	1
Inidian Hills						
MW 01	3/11/97	312	0	10.9	0	0
MW 10	3/11/97	358	0	1.16	0	0

Table 0-4. Results from four LUST monitoring wells in the East Mountain study areas.

1.9.3. Location of New Monitoring Wells

Using the prospecting analogy for finding ground water contamination, a strategy for locating new monitoring wells was used in which a single well was to be constructed in each study area, and information collected from this well would then be used to locate subsequent wells. As noted previously, the ideal scenario would consist of locating at least three wells in a water table aquifer of sufficient areal extent that the direction, rate of flow, and approximate extent of contamination could be determined. The objective of this approach was to measure the water quality in up gradient and down gradient wells, thus allowing determination of the impact of on-site systems on ground water quality beneath the subdivision. The strategy used in this study recognizes the uncertainty associated with finding either a water table formation of sufficient extent to allow completion of multiple monitoring wells or detecting degradation of ground water quality from on-site wastewater disposal systems. Accordingly, if data generated from the first monitoring well (i.e. lack of groundwater or lack of evidence of groundwater contamination) in each area suggests that a particular location will not achieve the objectives of this study another more suitable location would be sought.

The first phase of drilling involved installation of monitoring wells in Sandia Heights, Piñon Ridge, Sierra Vista, and Sandia Park. Figure 0-1 and Figure 0-4 show the location of the monitoring wells used in this study. The types of information collected from the well drilling and completion program included stratigraphy, ground water elevation, and aquifer hydraulic characteristics as determined by slug tests. Water samples from each of the monitoring wells were analyzed for the constituents listed in Table 0-5, and the drilling logs and static water levels were analyzed to determine whether water table aquifer conditions were present. In addition, water level measurements were used to validate the East Mountain water table contour map (Figure 0-4).

Two existing wells were found during a search of well records in the State Engineer's Office that fit the criteria for use as monitoring wells for this study. The first well is located near the intersection of Tramway Blvd and Paseo del Norte Blvd. Its location and construction make it suitable for use as a down gradient monitoring well, thus it was designated as SH01 and included in the network of wells sampled during this project. It is a private well that was constructed in 1979 by Robert Mhoon. The well has never been used and does not have a pump

installed. It meets the criteria of being screened across the water table. The well record gives a total depth of 520 feet with a water level after drilling of 455 feet. The depth to water is currently approximately 476 feet. Sounding of the well with a bailer shows that it has silted up to a depth of about 495 feet below ground surface. It is constructed of 5.5-inch steel casing with a screen interval of 40 feet at a depth ranging from 440 to 480 feet.

The second existing well of special interest that was identified in searching the State Engineer's records was drilled by the City of Albuquerque and is located in the Elena Gallegos picnic area. It is a domestic supply well and is currently used to provide water for the picnic ground and the caretaker's residence. The well is located approximately one mile east of the Sandia Heights subdivision and is completed in the Sandia Granite at a depth of 55 feet. This well is also screened across the water table. Its location and depth make it ideal for collecting uncontaminated background water samples. The well has an outlet for taking a sample directly from the well before any storage or treatment. It is designated as EG01. Permission collect water samples was obtained for both wells. Their use as monitoring wells saved approximately $30,000 in drilling costs. The well records are included in Appendices 1 and 4 of this report.

1.9.4.   Well Construction Methods

A wide variety of drilling methods have been used for constructing monitoring wells. These have been reviewed by the EPA (USEPA, 1993). Since ground water contamination from on-site disposal systems will first occur at the top of an underlying water table aquifer, it is necessary to identify the location of the top of the aquifer when constructing monitoring wells. This is an important factor to consider when selecting drilling methods. Drilling methods which use water-based drilling fluids (i.e. the mud rotary method) are therefore less desireable. However, the fact that the mud rotary method enables rapid drilling in almost all types of strata, including unconsolidated formations, makes it commonly used for installing monitoring wells.

Three drilling methods were considered for the Sandia Heights area: vibratory drilling, hollow stem auger, and mud rotary. The first two methods offer the advantages of using little or no drilling fluids, rapid penetration of unconsolidated formations, accurate identification of the top of the water table, and allow simple collection of drill cuttings which are representative of down-hole conditions to facilitate geologic logging. The vibratory or sonic drilling method has

been used for site investigations at Sandia National Laboratories and local UST siteswith some success. However, both auger and vibratory drilling methods suffer from the disadvantage of being unable to penetrate consolidated formations or large cobbles. Further disadvantages of the sonic drilling method is that it is expensive, the technology is not available locally, and is in an early stage of development, and drilling at depths greater than 200 feet becomes slow. A disadvantage of the hollow stem auger method is that it is generally limited to depths of 150 feet or less as the risk of losing the entire drill string increases at large depths. Based on an evaluation of these factors it was decided to use the mud rotary drilling method for constructing the Sandia Heights monitoring wells.

The geology of the East Mountain area is dominated by rock formations (principally limestone, shale, siltstone, and sandstone), hence, hard rock drilling techniques must be considered. Both air rotary and mud rotary drilling methods were considered in the East Mountain area. Consultation with drillers confirmed that the air rotary drilling method was appropriate for this area, hence that was the method requested in the well construction bid specifications. Copies of the bid specifications are listed in Appendix 2.

All drilling was supervised by a project hydrogeologist. Sample cuttings were taken every five feet during drilling. The samples were described and placed in sample bags and were archived with the New Mexico Bureau of Mines and Mineral Resources. Well construction details are adapted from the USEPA guidelines (USEPA, 1993) and are shown in Figure 0-5. PVC casing and screen material were used because it is inexpensive, easy to work with, and is compatible with the contaminants of interest in this investigation. Centralizers were used at the top and bottom of the screened interval to insure that the screen is placed in the center of the hole. All wells used 4 inch casing diameter. It was proposed that a 20 ft well screen be used in all wells up to 150 feet deep, and a 30 ft well screen be used in deeper wells, however, due to difficulties associated with locating the top of the water table in several of the wells, longer screen lengths were used in most of the wells as discussed below.

To protect the well from vandalism and prevent surface contamination from entering the well, a flush mounted steel surface casing and a concrete pad were constructed. A locking steel cap was provided. A coarse filter pack of 20 - 40 mesh quartz sand was placed along the entire length of well screen by a tremie pipe in each well.   A sand bridge was placed on top of the filter

back. Bentonite pellets were placed above the sand bridge, and the annulus between the wall of the hole and the casing were backfilled with a cement-bentonite grout. The cement-bentonite backfill was placed by tremie pipe to assure complete filling of the annular space.

Well development was accomplished strictly by pumping each well. No hydraulic jetting or other methods of development which might introduce additional water to the formation were used. Slug tests were performed following completion of each well to allow estimation of the formation's hydraulic conductivity. The limitations of this testing procedure are well known, however, it is a simple and inexpensive test that does provide some information about the aquifer.

A bid package was prepared (Appendix 2) and bids were received in April of 1997. Rodgers & Co. Inc, from Albuquerque, NM was selected as the lowest bidder and contracted to drill the monitoring wells. Drilling of the first five wells began in August of 1997. Rodgers & Co. used a Gardner-Denver mud rotary rig to drill eight inch holes for the two wells in Sandia Heights (SH02, SH03). Each well took three days to drill and two days to install the well casing and screen, and develop. The wells were developed by pumping approximately 1500 gallons of water in two stages over a period of three hours. The first stage of development was done at 10 gpm for 1.5 hours until the water was clear. The well was allowed to sit for half an hour and then for the second stage of development it was pumped with a portable down-hole pump at the maximum rate until breaking suction, and then the rate was backed off to 10 gpm, and pumping continued until the water was clear of sediments. The wells each produce a maximum flow of about 11gpm without breaking suction at the pump.

FLUSH WELL COVER

5 FT DEEP CONCRETE SUPPORT

CEMENT/BENTONITE BACKFILL

PVC WELL CASING 4 IN. DIAMETER

BENTONITE PLUG 5 FT LENGTH

SAND BRIDGE 5 FT LENGTH

CENTRALIZERS AT TOP & BOTTOM OF WELL SCREEN

GRAVEL FILTER PACK

WATER TABLE

PVC WELL SCREEN UP TO 200 FT. TOTAL LENGTH

5 FT SEDIMENT SUMP
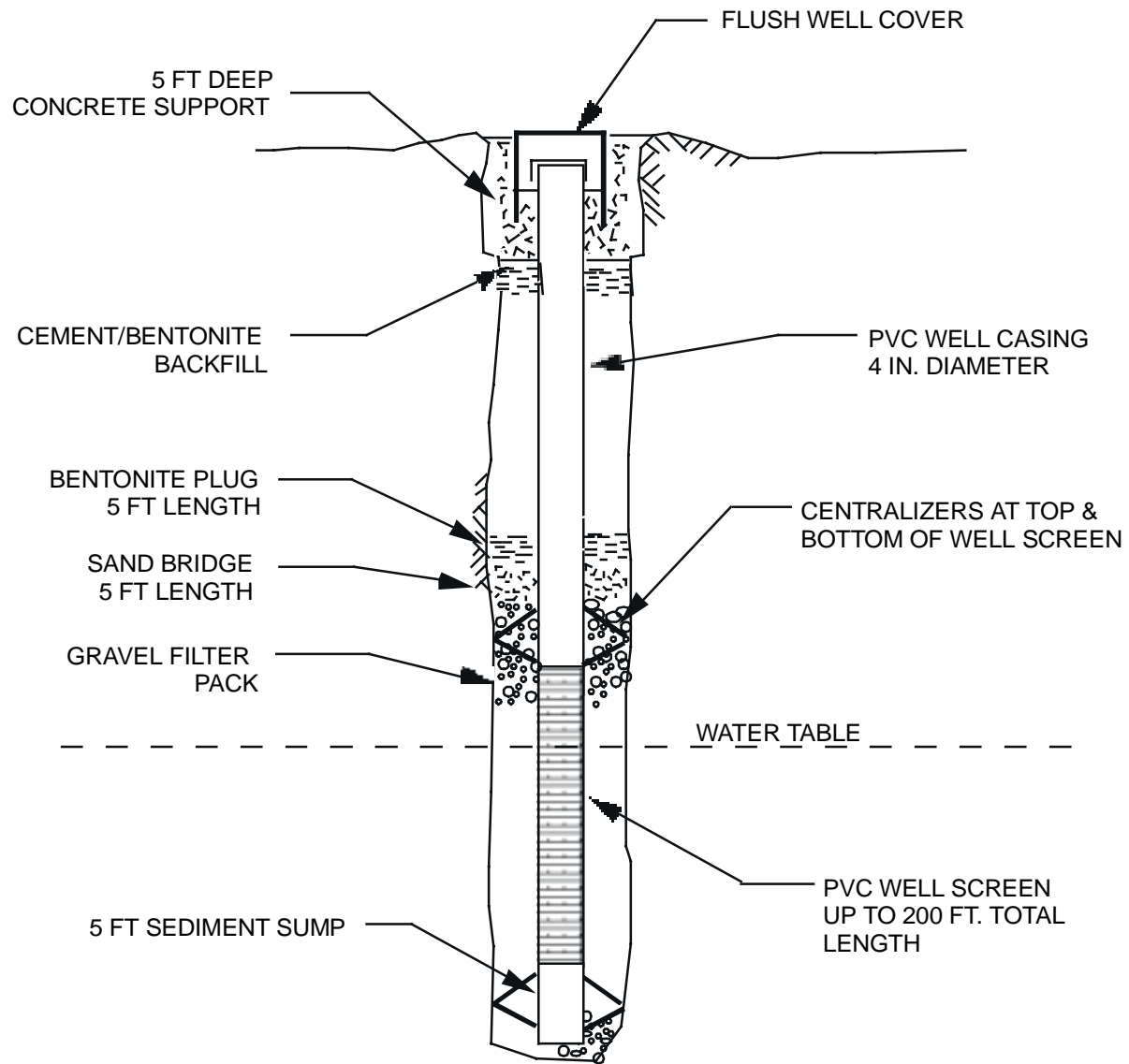
Monitoring well construction details

Figure 0-5. Monitoring well construction details.

The three East Mountain wells were also drilled by Rodgers & Co. using an Ingersoll-Rand TH100 air rotary rig. The Piñon Ridge well (PR01) was drilled first, using an air foam mixture in an eight inch hole. The hole was drilled to a depth of 220 ft the first day. Water was found in the hole the following morning and the water level in the well was still rising at this time requiring a decision as to where to locate the well screen. Because of the slow recovery rate and the use of air-foam drilling fluid the exact depth to ground water was not known. Because of this uncertainty, and the need to screen across the water table, 160 ft of screen was installed in the well. This well was completed entirely in the Mancos Shale. The well was developed with a portable downhole pump. This was repeated twice and then the well was pumped at its natural rate of 0.25 gpm for another hour. A total of 325 gallons was pumped from the well which included water from the water truck.

The Sierra Vista well (SV01) was drilled with a hole diameter of 6 inches using air without foam as the drilling fluid. The first day of drilling was completed to 120 ft. On the second morning the well was dry and drilling continued. Drilling stopped on the second day at 340 ft. On the third morning water was measured at 283 feet in the hole. The drill string was pulled out and the water elevation in the well was observed to rise from 329 ft to 325 ft at a steady rate throughout the afternoon. The next morning the water level was 309 ft and still rising. 220 feet of screen was placed from 340 to 120 feet. Due to the very low ground water flow rate into the well, development was performed by bailing. A total of about 75 gallons of water was removed from the well.

Sandia Park well number 1 (SP01) took two days to complete by drilling with dry air in a six inch hole. This well was completed in the Chinle Formation to 110 ft depth in a confined sandstone aquifer that was encountered at 95 ft and produces at least 8 gpm of water. The static water level is 54 feet, thus it is apparent that this strata constitutes a confined aquifer. The well was developed by airlift pumping an estimated 450 - 500 gallons from the well. Sandia Park well number 2 (SP02) was completed in four days to a depth of 180 feet in the Yeso Formation by drilling a six inch hole using air as the drilling fluid. A confined gravel/coarse sand layer that produced water was encountered at ~160 feet depth in the Yeso formation. The static water level was established at 130 feet below ground surface. This well was developed by airlift pumping

approximately 2400 gallons of water from the well. The presence of confined water bearing formations at such shallow depths in the Sandia Park area was not expected. This may affect the results of the ground water sampling program as the confining zone may prevent or delay contaminants from on-site disposal systems from reaching the underlying aquifer.

A dry monitoring well was drilled on the western boundary of the Sandia Knolls subdivision. This site was picked because it appeared to be at the toe of a surface drainage basin, and the UNM team speculated that ground water flow followed the surface topography. Drilling was completed to 280 feet, but no groundwater was reached (no water was in the hole after standing 16 hours), and the hole was abandoned and backfilled from the surface with drill cuttings.

Final static water levels and as-built diagrams for all of the monitoring wells are shown in the figures below. Copies of the field logs including description of the cuttings for all six wells and the dry borehole in the Sandia Knolls subdivision are contained in Appendix 5.

Diagrams of all of the wells constructed for this project are contained in Figure 0-6 through Figure 0-11. These figures show the screened interval and a description of the lithology as determined by examination of the drill cuttings.

Cedar Hill Well (SH02)

Elevation 6075 ft

GW

GW - Well graded sandy gravels with approximately equal amounts of K-spar, plagioclase, quartz, and minor mafics and limestone

85

SW 95

SW - Moderately graded sands and gravely sands with approximately equal amounts of K-spar, plagioclase, quartz, and minor mafics and limestone

165

SW

190

GW

255

SW

440 ft - Top of Screen

WT= 443 ft

0   100 FT

495 ft - Bottom of Screen

TD-500 ft

Figure 0-6. Diagram and lithology of the Cedar Hill (SH02) monitoring well in the Sandia Heights subdivision.

San Rafael Well (SH03)

Elevation 6050 ft

SW

GP

GW - Well graded sandy gravels with approximately equal amounts of K-spar, plagioclase, quartz, and minor mafics and limestone

SW

GP

SW - Moderately graded sands and gravely sands with approximately equal amounts of K-spar, plagioclase, quartz, and minor mafics and limestone

GP - Poorly graded gravels with approximately equal amounts of K-spar, plagioclase, quartz, and minor mafics and limestone

SW

GW

SW

440 ft - Top of Screen
WT= 444.71 ft

0   100 FT

485 ft - Bottom of Screen
TD-490 ft

Figure 0-7. Diagram and lithology of the San Rafael (SH03) monitoring well in the Sandia Heights subdivision.

Piñon Ridge Well (PR01)

Elevation 6990 ft

Clay

Fine to very fine grained sandy siltstone, thinly layered, lightly indurated

Fine grained silty sandstone, thinly layered, moderately indurated

120 ft - Top of Screen

Medium to Fine grained, organic rich, silty sandstone, thinly layered, moderately indurated

WT= 173.80 ft

Varve deposits, 2-3 mm thick layers

215 ft - Bottom of Screen
TD-220 ft

0   100 FT
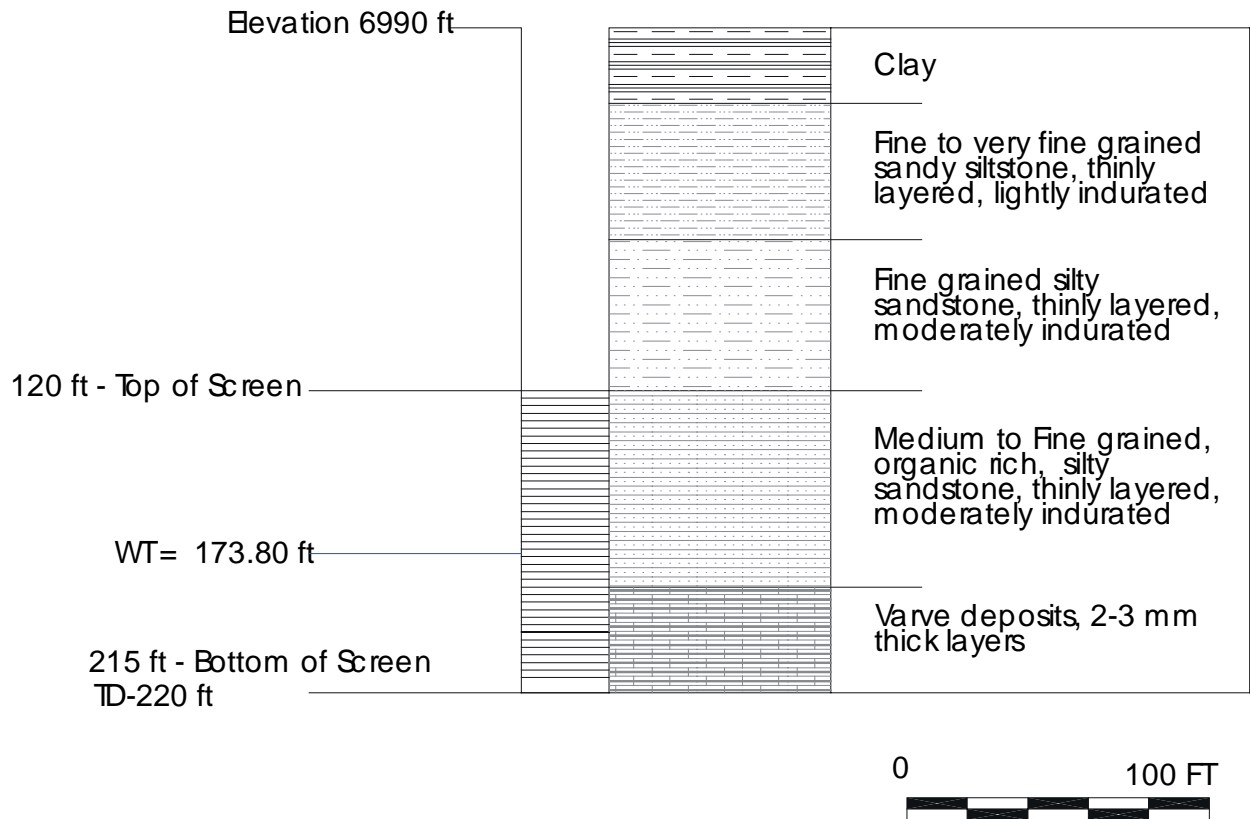
Figure 0-8. Diagram and lithology of the Piñon Ridge monitoring well (PR01) in the East Mountain area.

Sierra Vista Well (SV01)

Elevation 6960 ft

Fine- to v. fine sandstone

Clay rich siltstone

Sandy siltstone

Very fine silty sandstone

Mudstone

120 ft - Top of Screen

Fine- to v. fine silty sandstone

WT= 239.50 ft

Gravelly sandstone

Fine- to v. fine silty sandstone

335 ft - Bottom of Screen
TD-340 ft

0   100 FT
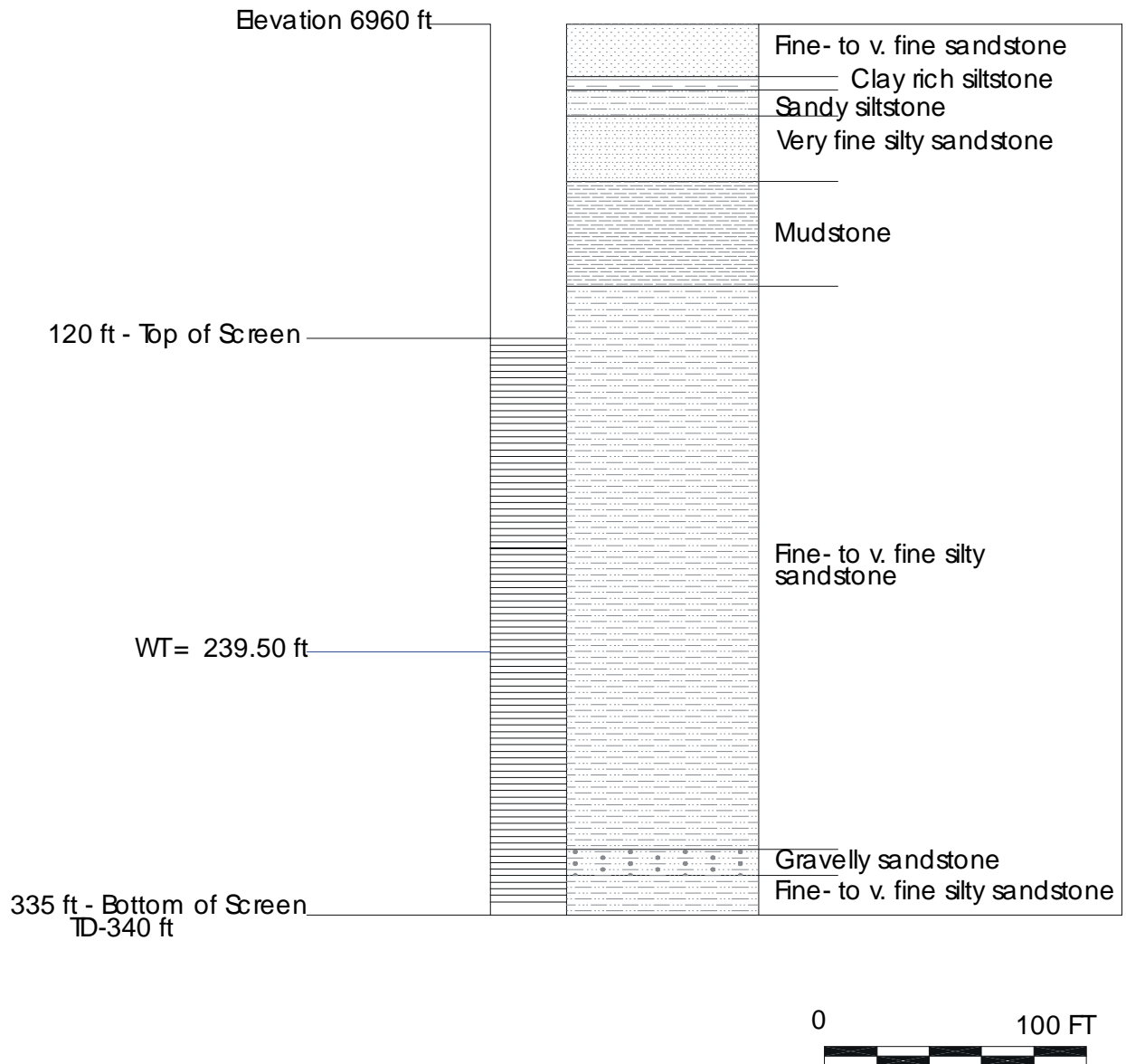
Figure 0-9. Diagram and lithology of the Sierra Vista monitoring well (SV01) in the East Mountain area.

Sandia Park Well 1 (SP01)

Elevation 6860 ft

Siltstone

Sandy siltstone

Very fine sandstone

WT= 55.00 ft

Mudstone

120 ft - Top of Screen

105 ft - Bottom of Screen

Very fine sandstone

TD-110 ft

Mudstone

0   100 FT
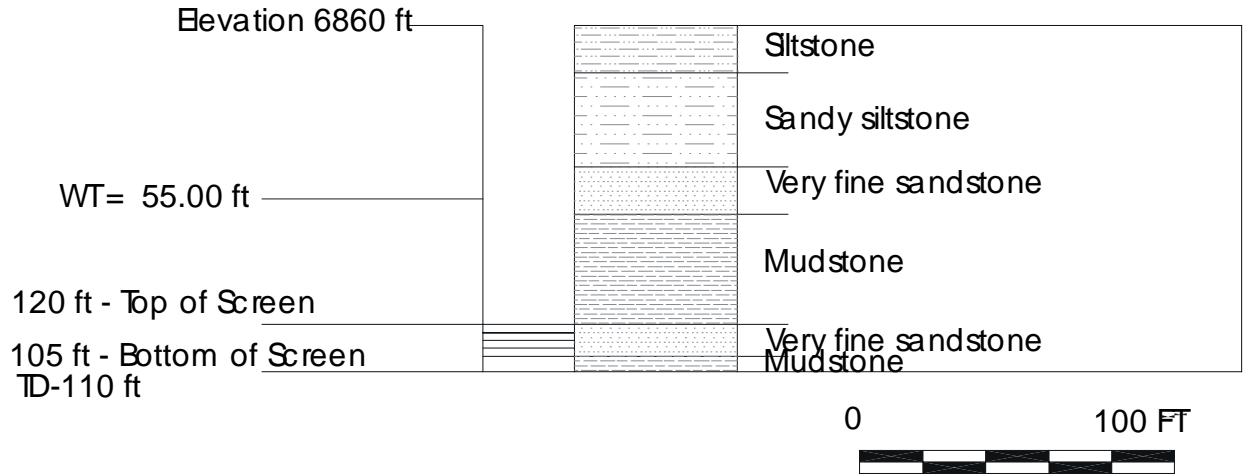
Figure 0-10. Diagram and lithology of the Sandia Park monitoring well no. 1 (SP01) in the East Mountain area.

Sandia Park Well 2 (SP02)

Elevation 6935 ft

Alluvial deposits of poorly sorted gravels/sands/fines/ composed of quartz sandstones arkoses, limestones near the surface and well sorted gravels and medium to fine sands of the same composition below 15 ft

Glorieta Sandstone - well sorted, fine-grained, well indurated quartz arenite

Yeso Formation - moderately consolidated fine-grained to very fine-grained mudstones, siltstones, coarse sands, ranges from calcareous to non-calcareous

WT= 173.80 ft

155 ft - Top of Screen

175 ft - Bottom of Screen

Unconsolidated coarse sand and gravel of arkose, quartz, limestones with minor dark minerals too small to identify with hand lens

TD-180 ft

0   100 FT

Figure 0-11. Diagram and lithology of the Sandia Park monitoring well no. 2 (SP02) in the East Mountain area.

60

1.9.5. Ground Water Sampling, Preservation & Analysis

Four rounds of quarterly ground water sampling were performed starting in December, 1997 and ending in September, 1998. Sampling of six of the project's seven monitoring wells was done with a submersible Bennett pump powered with nitrogen gas. The Sandia Heights upgradient well is a domestic well that has a pump installed. This well was purged at ~ 8 gpm for 5 minutes, then samples were taken directly from the well before any storage or treatment. The other six wells were purged at ~ 1.5 gpm of approximately three bore volumes of water when possible. Temperature, conductivity, pH, and DO were measured during the purging and usually stabilized after 10 - 15 gallons (or less) were purged. The Sierra Vista and Piñon Ridge wells produced less than 1/4 gpm and were pumped dry, allowed to recover, and then pumped as much as time allowed before sampling. With each sampling round the Piñon Ridge well produced slightly more water until after four rounds, the well was producing enough water to purge the full three bore volumes at a flow rate of ~ 1.5 gpm. The Bennett pump was decontaminated after sampling each well with 8 - 10 gallons of deionized (DI) water with Liquinox detergent and then approximately 10 - 15 gallons of DI water was pumped through the system. Also, the hose reel was rinsed with DI water. Chain-of-custody forms were kept for each sample from the point of sample collection to the City Water Quality laboratory. Table 0-5 gives the sample type, type of container and preservation methods, and analytical methods used for each sample.

Sample Type	Container	Preservation	Method

Bacteriological
    E. coli
    Total coliforms
	125 mL / Plastic	Pre-preserved with
		Sodium Thiosulfate
			18SM9215C
			WQLMUG01

General Chemistry
    Alkalinity
    Ammonia
    Bromide
    Bicarbonate
    Carbonate
    Chloride
    Fluoride
    MBAS
    Conductivity
    Sulfate
    Total Diss. Solids
	1000 mL / Plastic	Refrigeration	18SM18SM2320B.4
			18M4500NH3E
			18SM4110B
			18SM2320B.5.3
			18SM2320B.5.1
			18SM4110B
			18SM4500F.C
			Calc.
			18SM2510
			18SM4110B
			18SM2540C

Metals
    Calcium
    Iron
    Magnesium
    Potassium
    Sodium
    Zinc
	200 mL / Plastic	Nitric Acid
		to pH < 2.5
			18SM3120B
			18SM3120B
			18SM3120B
			18SM3120B
			18SM3120B
			18SM3120B

Total Organic Carbon	250 mL/ Amber Glass	Sulfuric Acid to pH < 2.5	18SM5310C

Nutrients/Demands
    Nitrate
    Nitrite
    O-Phosphate
    TKN
	250 mL/ Plastic	Sulfuric Acid
		to pH < 2.5
			18SM4500NO3F
			18SM4500NO3F
			18SM4500P..F
			18SM4500NORGB

Table 0-5. List of analytes and sample preservation methods.

A suite of field parameters was measured, including pH, temperature, dissolved oxygen and electrical conductivity. These are inexpensive measurements to perform and assist in understanding solution chemistry. Similarly, laboratory measurements of pH, alkalinity, electrical conductivity, and major ion analyses are important indicators of general water quality as the major ion concentrations dominate the chemical behavior of the solution and can be used to compare the characteristics of water produced from different wells. Analysis of the nitrogen species is important because nitrate is the principal ground water contaminant of concern from

on-site systems. In addition, analysis for the other nitrogen species, as well as analyses of reduced iron (Fe(II)), manganese (Mn(II)) and sulfide (HS⁻) ions permits a further evaluation of the oxidation-reduction (redox) chemistry of each water sample. Analysis of coliform bacteria provides an indication of whether direct contamination of ground water by poorly filtered wastewater has occurred. Analysis of total organic carbon (TOC) allows determination of the available energy source for heterotrophic microbial activity in the underlying soil/water column, and is also an important indicator of redox chemistry of the solution.

Six of the monitoring wells were sampled by the USGS and analyzed for 66 organic compounds associated with domestic wastewater. The wells sampled and the dates they were sampled are: EG01, 7/1/98; SH03, 7/1/98; PR01, 6/18/98; SV01, 6/17/98; SP01, 6/18/98; and SP02, 12/3/98. Table 0-6 lists these compounds and identifies their potential use or how they might come to be present in groundwater.

Compound Name	Use	Compound Name	Use
tetrachloroethylene	Solvent, degreaser	methyl parathion	Pesticide
tribromomethane	Trihalomethane	carbaryl	Pesticide
benzaldehyde	Flavor	Nonylphenol ethoxylate 1 (NPEO1) (total)	Nonionic detergent metabolite
phenol	General disinfectant	metolachlor	Pesticide
1,2,4-trimethylbenzene	Common in gasoline	chlorpyrifos	Pesticide
1,3-dichlorobenzene	Fumigant	DCPA	Pesticide
1,4-dichlorobenzene	Fumigant	fluoranthene	PAH
D-limone	Fragrance	trans-chlordane	Pesticide
1,2-dichlorobenzene	Fumigant	pyrene	PAH
acetophenone	Fragrance	cis-chlordane	Pesticide
para-cresol	Wood preservative	bisphenol A	Used to manufacture polymers
ethanol, 2-(2-butoxyethoxy-)	Solvent for nitrocellulose resins	p,p' -DDE	Pesticide
naphthalene	hydrocarbon (PAH), fumigant	dieldrin	Pesticide
naphthalene, 2-methyl	PAH	NPEO2-total	Nonionic detergent metabolite
naphthalene, 1-methyl	PAH	tri (dichloroisopropyl) phosphate	Fire retardant
phthalic anhydride	Used to manufacture plastics	bis (2-ethyl hexyl) adipate	Plasticizer
2(2-btxyethxy) ethyl acetate	Solvent for nitrocellulose resins	ethanol, 2-butoxy-, phosphate	Plasticizer
2,6-di-tert-butylphenol	Antioxidant	trephenyl phosphate	Plasticizer
acenaphthylene	PAH	benzo (a) anthracene	PAH
2,6-di-tert-para-benzoquinone	Antioxidant	chrysene	PAH
Butylated hydroxyanisole (BHA)	Antioxidant	bis (2-ethyl hexyl) phthalate	Plasticizer
acenaphthene	PAH	17beta-estradiol	Major estrogen metabolite
Butylated hydroxytoluene (BHT)	Antioxidant	benzo (b) fluoranthene	PAH
fluorene	PAH	benzo (k) fluoranthene	PAH
tributyl phosphate	Plasticizer	benzo (a) pyrene	PAH
nonylphenol-total	Nonionic detergent metabolite	3 beta-coprostanol	Carnivore fecal indicator
simazine	Pesticide	cholesterol	Fecal indicator
atrazine	Pesticide	indeno (1,2,3-c,d) pyrene	PAH
tri (2-chloroethyl) phosphate	Fire retardant	dibenzo (a,h) anthracene	PAH
lindane	Pesticide	benzo (g,h,l) perylene	PAH
phenanthrene	PAH	methyl-di-(tert-butyl-d9) phenol-SURROGATE (BHT-d9)	Monitor method performance
diazinon	Pesticide	nonylbenzene - SURROGATE	Monitor method performance
anthracene	PAH	bisphenol A,-d6 SURROGATE	Monitor method performance
caffeine	Stimulant		

Table 0-6. List of compounds analyzed by USGS.

1.10.   Pollutant Transport and Transformation in the Vadose Zone

1.10.1. Objectives

A better understanding of the contaminant transport and transformation phenomena which occur beneath on-site liquid waste disposal systems is necessary to develop methods of predicting ground water contamination from on-site treatment and disposal systems. Knowledge of vadose zone transport and transformations also allow an improved understanding and interpretation of the results of the ground water monitoring program.

Movement of water and contaminants in the soil beneath an absorption field is controlled in large part by the hydraulic characteristics of the unsaturated soil. In particular, the rate at which the effluent migrates downward toward the ground water is important. Because many of the reactions and transformations which control the contaminant speciation and concentration are time-dependent, the rate of vertical movement dictates how much time there is for these reactions and transformations to occur. Once the pollutants reach the water table the nature and rate of the transformation reactions changes dramatically primarily because oxygen transport is orders of magnitude slower in ground water than in the vadose zone. Consequently, an understanding of the transport characteristics of the underlying materials will provide key information to the understanding of the fate of on-site disposal system effluent and the final quality of the underlying ground water.

Septic tank effluent which flows into the unsaturated soils of the drain field contains high concentrations of organic nitrogen, ammonium ion (NH_4^+) and biodegradable organic compounds.   The concentrations of these compounds and the types of microbial reactions which may occur are discussed in section 2.1.

The vadose zone investigation was conducted to determine the contaminant transport and transformation through unsaturated soils at a site representative of conditions on the west side of the Sandia Mountains where depth to ground water was in excess of 100 ft. It was accomplished by conducting a series of soil borings in a thick unsaturated formation beneath a relatively new absorption field system. Core samples were collected and analyzed for constituents that could be used to determine the rate of water flow through the soil, and would be indicative of the microbial transformations occurring within.

1.10.2. Site Description

The sampling site was chosen to be in thick, unsaturated and unconsolidated Santa Fe deposits. Furthermore, its location in North Albuquerque Acres is representative of the geohydrologic conditions that are present in an unsewered area of Bernalillo County that is experiencing rapid and intensive development. Based on its density and the high permeability of the soils, this development appears to be a threat to underlying ground water quality. The original intent of this phase of the project was to sample up to five locations, however, due to concerns about liability associated with drilling operations at private residences, the sampling program was limited to one site which in turn was investigated more thoroughly. The site chosen was the Bernalillo County Sheriff's Substation located on Paseo del Norte. This facility, constructed in 1997 is operated 24 hours a day. It includes showers, a small kitchen, and bathrooms, hence it produces wastewater that is similar in composition to domestic wastewater; in particular there are no industrial waste components. The wastewater treatment system consists of a septic tank and two absorption fields, each consisting of a single perforated pipe approximately 100 ft long. A selector valve was provided so that the flow could be sent to either of the two pipes. The UNM project team was informed that only one leach line had been operated since construction of the system. Because there is negligible irrigated landscaping it is assumed that all of this water eventually flowed into the septic tank-absorption field system.

A diagram of the system is presented in Figure 0-12. The absorption field is located on an undeveloped part of the property adjacent to a paved parking lot. Five bore holes were drilled at this site, each to a depth of 65 feet, and their locations are shown in the figure.

Figure 0-12. Diagram of the Bernalillo County Sheriff's Substation absorption field system showing location of soil sampling bore holes.

Parking Lot

Primary Absorption Field

Hole C

Hole D

Hole E

5 ft

25 ft

25 ft

25 ft

Selector Valve

10 ft

Alternate Absorption Field (unused)

25 ft

0 '   10'   20'

Hole B

42 ft

Hole A
(Background)

67
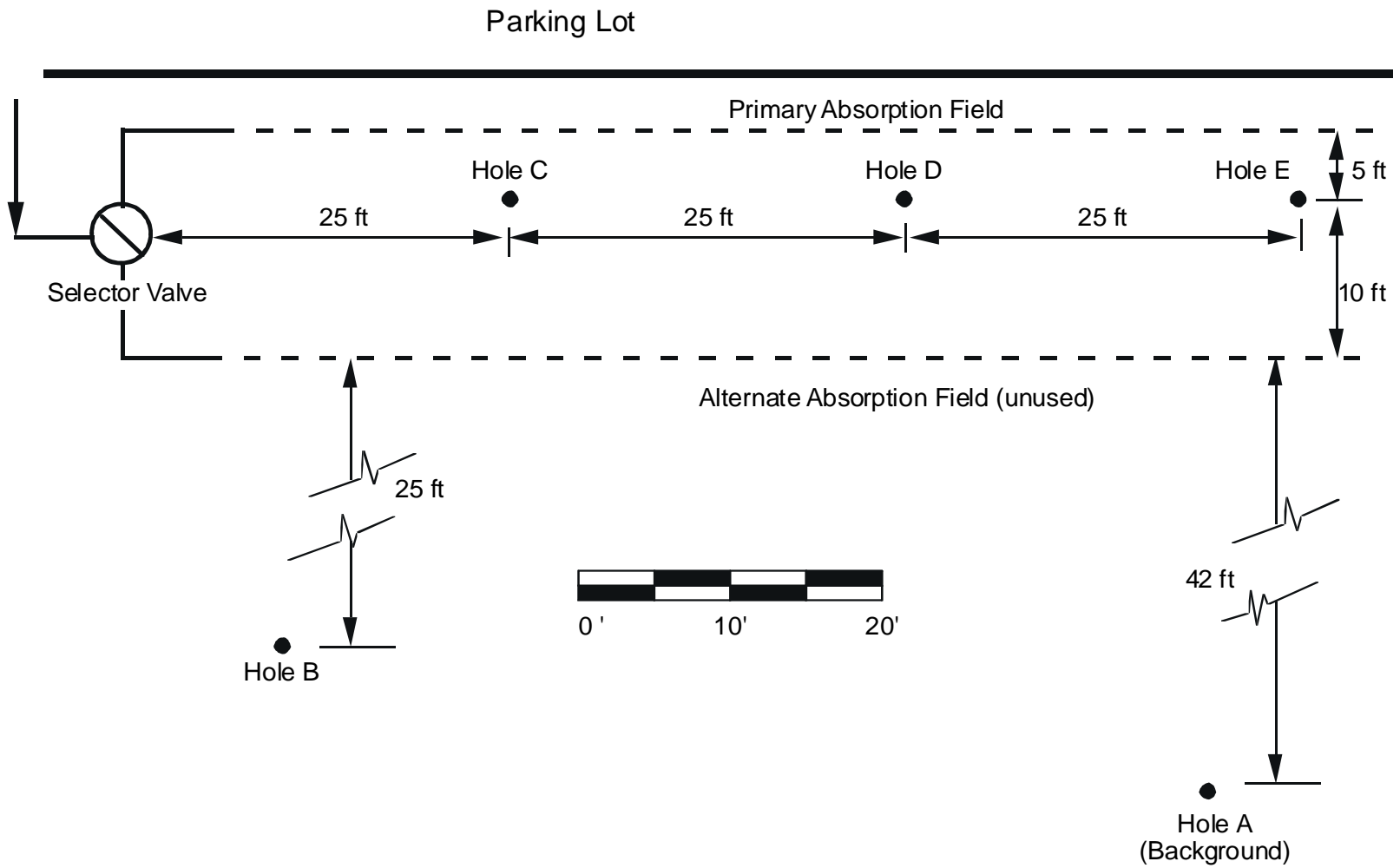
1.10.3. Sampling Methods

To produce the most representative results a soil sampling program was designed to collect relatively undisturbed samples of the unconsolidated soil at the site. This was accomplished by drilling with a hollow-stem auger and using a split-spoon sampler to collect the soil samples. A drilling contractor, SHIB, Inc. of Sandia Park, NM, was selected based on a competitive bid process. Drilling was accomplished using a 5 ft long hollow stem auger flights, with an outer diameter of 6 1/4 inches and a two inch diameter barrel. A top drive drill rig was used to power the auger.

The hollow-stem sampler consisted of two barrel halves, a drive shoe, and a sampler head containing a ball check valve, all of which were threaded together. The sampler had an inside diameter of 1.5 inches. The sampler was threaded onto drilling rods and lowered to the bottom of the boring through the barrel of the hollow-stem auger flights. The sampler was then driven into the soil with blows from a drop hammer attached to the drill rig. The sampler was driven out 1 to 2 feet ahead of the drill bit then extracted from the soil by upward blows with the drop hammer (ASTM, 1991 c). The sampler was then retrieved and the split spoon was opened to remove the sample.

At each of the bore holes, samples were collected at depths of: 1 ft, 3 ft, 5 ft, 10 ft, 15 ft, and thereafter at 5 foot intervals up to a maximum depth of 60 feet. Each sample was placed in an airtight container and returned to the University of New Mexico for laboratory analysis. The bore holes were backfilled with a 10% mixture of dry bentonite chips and native soil to prevent potential vertical migration of surface contaminants through high conductivity material associated with unconsolidated backfill. It was felt that due to the relatively large diameter of the holes and their shallow depth, a tremie pipe was not needed to place the backfill, thus the mixture was poured into each hole from the surface as the auger was removed from each hole.

1.10.4. Sample Analysis

Six different geotechnical tests were performed on the samples to determine the physical properties of the soil. All of the tests were performed in the Civil Engineering laboratories at the

University of New Mexico. Due to the high hydraulic conductivity of the soils at this site it was believed that most of the septage would infiltrate near the beginning of the absorption field, therefore, soil samples from borehole "C" would be expected to show the greatest influence from the nearby absorption field. Thus, the bulk of these measurements were focused on samples from borehole "C". Table 0-7 lists the tests performed on these soil samples.

PARAMETER	METHOD
Water Content Determination	ASTM D 2216-90
Particle Size Analysis Mechanical Method	ASTM D 421, Sample Procedure ASTM D 422, Test Procedure
Atterberg Limits	ASTM D 4318
Specific Gravity	ASTM D 854
Hydraulic Conductivity	ASTM D 2434-68 Falling Head Method
Moisture Characteristic Moisture Characteristic Curve	Procedure developed at UNM

Table 0-7. Geotechnical tests conducted on soil samples recovered from the vadose zone investigation.

A portion of the core material from the split-spoon core barrel was placed in a plastic bag, air-dried, then stored for later analysis of grain size. The rest of the core material was used for determination of moisture content, moisture characteristic, hydraulic conductivity and for selected chemical and bacterial analyses.

Gravimetric moisture content (w) was determined by comparing the weight of the water in the sample (M_w) to the weight of the solids in the sample (M_s) (oven dried at $105\,^{o}$ C for 24 hours).

$$w = \left(M_w \middle/ M_s \right) * 100 \tag{3-1}$$

This test was performed on every sample retrieved within 24 hours of recovery.

Soil classification was determined based on the grain size analyses and the Atterberg tests.

Standard sieves were used for the sand and gravel fractions. A hydrometer analysis of silt and

clay sized particles was not necessary due to the coarseness of the materials. Results of these analyses, for hole "C", are given in Table 0-8.

Sample Depth (ft)	Water Content	Ks (cm/sec)	Unified Soil Classification
15	9.54	3.92×10^{-3}	Silty sands
20	5.88	3.92×10^{-3}	Well graded sands, silty sands
25	6.26	2.36×10^{-1}	Silty Sands
30	4.2	2.28×10^{-1}	Poorly graded sands
35	11.44	1.3×10^{-3}	Clayey sands
40	12.22	4.77×10^{-3}	Clayey sands, silty sands
45	14.55	1.3×10^{-3}	Clayey sands
50	11.13	3.92×10^{-3}	Silty Sands
55	9.73	2.36×10^{-1}	Well graded sands
60	14.83	3.92×10^{-3}	Silty sands
65	12.58	4.77×10^{-3}	Clayey sands, silty sands

Table 0-8. Summary of water content, hydraulic conductivity and soil classification for bore hole C.

The specific gravity of the soil is the ratio of the unit weight of solids (Ws / Vs) in the soil to the density water (ρ_{H2O}). In equation form,

$$G_s = \frac{W_s}{(V_s * \rho_{H2O})}$$
(3-2)

The results from this test were used later in the permeability experiments.

The saturated hydraulic conductivity (K_s) measures the ability of a porous media to transmit water. The hydraulic conductivity is usually defined in terms of Darcy's Law where the flow rate (Q) of a fluid through a soil of cross-sectional area A is proportional to the potential gradient (i). K_s is the proportionality constant:

$$Q = K_s iA$$
(3-3)

The water content profile within a soil can be determined from the suction head profile using a moisture characteristic function that relates the volumetric water content (Θ_v) to the suction head, i.e.

$$\Theta_v = \Theta_v(h) \tag{3-4}$$

where $\Theta_v(h)$ is the volumetric water content at suction head h. This function is obtained by conducting a hanging column experiment. The experiment is set up as shown in Figure 0-13. To obtain a drying curve the bottle is raised above the soil sample to induce a positive pressure head. Once the sample is saturated the bottle is lowered so that the water level in the bottle is even with the sample and suction head (h) is equal to zero. The apparatus is left in this position until equilibrium is reached (approximately 1 to 2 days). Then the sample is removed and weighed. The water bottle is then lowered in increments of 15 cm and the processed is repeated until the head difference is 150 cm.

At the conclusion of the test a curve is fit to the data to obtain the fitting parameters (α, n, and m) which are used in van Genuchten's function for moisture content ($\Theta(h)$), and unsaturated hydraulic conductivity ($K(h)$) as a function of the soil water potential head (h) the residual moisture content ($\theta_r(h)$), and the saturated moisture content ($\theta_s(h)$).

$$\Theta(h) = \Theta_r + (\Theta_s - \Theta_r)\left[1 + (\alpha h)^n\right]^{-m} \tag{3-5}$$

$$K(h) = K_s \frac{\left\{1 - (\alpha h)^{mn}\left[1 + (\alpha h)^n\right]^{-m}\right\}^2}{\left[1 + (\alpha h)^n\right]^{ml}} \tag{3-6}$$

where:

    h = suction head

    $\Theta(h)$ = volumetric moisture content as a function of suction head

    Θ_r = residual moisture content where adsorptive forces are dominant

    Θ_s = moisture content under saturated conditions

K(h) = hydraulic conductivity as a function of suction head

K$_s$ = hydraulic conductivity under saturated conditions

α, n, m = empirical constants

The results of these tests are used in the vadose zone water transport model and are presented in Appendix 7.
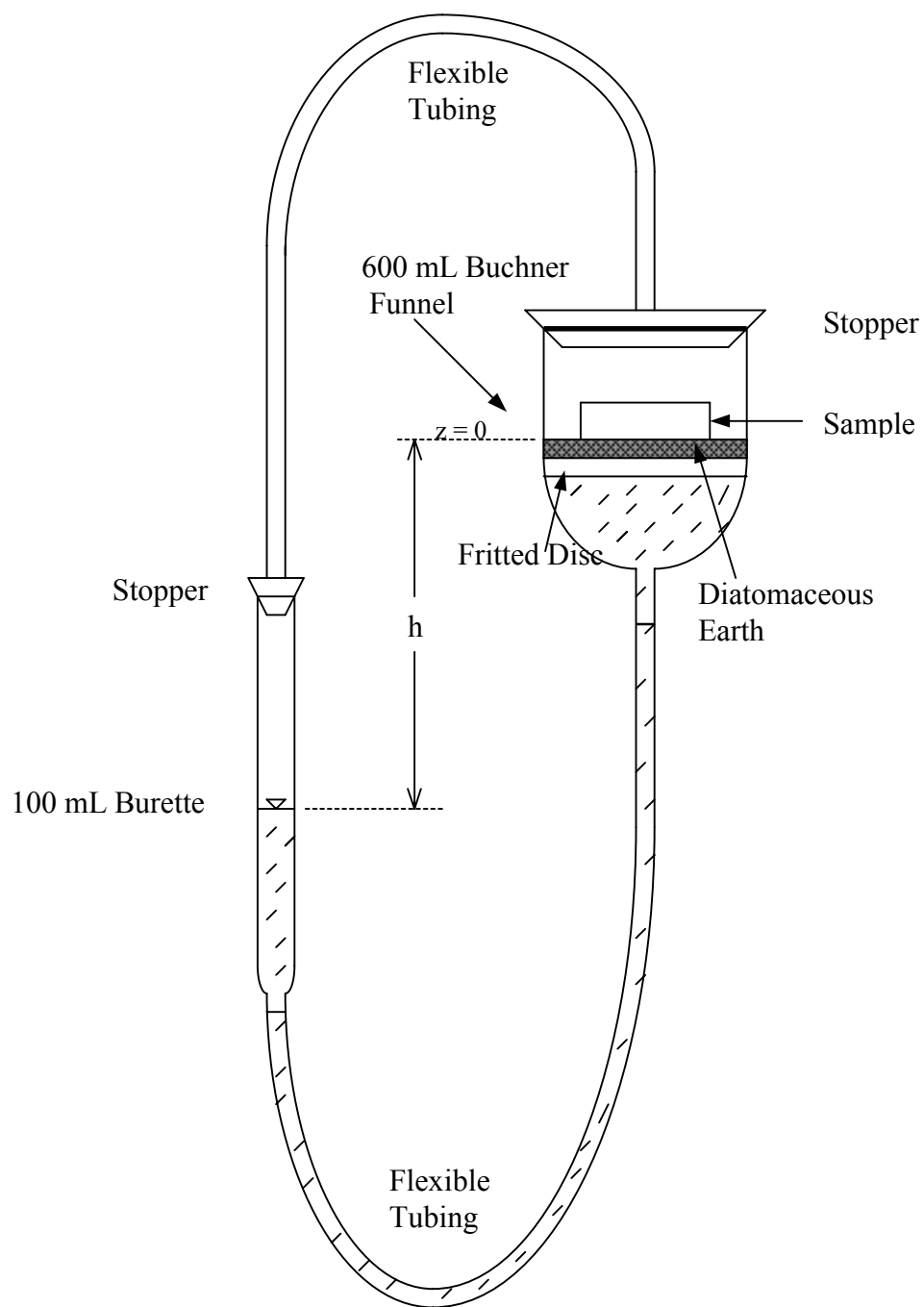
Figure 0-13. Diagram of the hanging column apparatus.

The main focus of the chemical analyses was obtaining information regarding the transport and transformations of nitrogen species in the soil column. Chemical oxygen demand (COD), soil pH, nitrate, total nitrogen, ammonia, and a suite of anion concentrations were measured. The geochemical tests, with the exception of ammonia, were performed on all of the samples recovered from the 5 holes. The ammonia test was performed on a smaller number of samples, due to the increased cost of these analyses and the finding that most soil samples had little or no ammonia present. Table 0-9 contains a complete list of the tests performed and the methods that were used.

PARAMETER	METHOD
Ammonia (NH_3)	HACH Test 'n' Tube procedure
Total Nitrogen (TKN)	HACH Test 'n' Tube procedure
Nitrite (NO_2^-)	Standard Methods 4500- NO_2^-.C Ion Chromatographic Method
Nitrate (NO_3^-)	Standard Methods 4500- NO_3^-.C Ion Chromatographic Method
Chemical Oxygen Demand (COD)	Test procedure followed as described in HACH COD reactor manual
Phosphate (PO_4^{3-})	Standard Methods, 4110 B Ion Chromatographic Method
Chloride (Cl^-)	Standard Methods, 4110 B Ion Chromatographic Method
Sulfate (SO_4^{2-})	Standard Methods, 4110 B Ion Chromatographic Method
Bromide (Br^-)	Standard Methods, 4110 B Ion Chromatographic Method
Soil pH	ASTM D 4972-89

Table 0-9. Chemical analyses performed on bore hole soil samples.

The chemical analyses of the pore water contained in the soil samples was determined by leaching each sample with deionized water. The leachate solutions were prepared by adding 250 mL of deionized water to approximately 50 g (wet weight) of material. The 5:1 mass ratio of

water to soil was used to provide enough water for conducting the chemical analyses, but not too much water to cause dilution of the constituents below detectable concentrations. The resulting leachate contains diluted pore water plus any soluble salts and exchangeable ions present on the soil samples.

The soil leaching results are presented as pore water concentration values in Appendix 7. The pore water concentrations were computed based on the measured concentrations in the extract solution using the equation:

$$C_p = C_e \frac{M_w}{M_s w \rho_{H2O}}$$

(3-7)

where:

C_p is the concentration of the constituent in the soil pore water (mL)

C_e = the measured concentration in the deionized water leachate solution (mL)

M_w = the mass of the deionized water used in the extraction procedure (g)

M_s = the mass of the soil sample used in the extraction (g)

w = gravimetric water content of the soil (g/g)

ρ_{H2O} = density of water (1.0 g/mL)

The large dilution that results during extraction leads to substantial uncertainty in the calculated pore-water concentrations, especially for those samples with very low moisture content. The data for nitrate that is given in this report includes the analysis of nitrate plus nitrite. However, because nitrite was present in negligible quantities, the results are referred to simply as nitrate.

All microbiological analyses were done in the Civil Engineering laboratories at UNM. The samples used for these analyses were recovered from borehole "C", again, due to its close proximity to the head of the absorption drain field line. The microbial tests that were performed are listed in Table 0-10.

PARAMETER	METHOD USED
Total Heterotrophs	Standard Methods 9215 Heterotrophic Plate Count
Total Coliform Bacteria	Standard Methods 9222 B. Standard Total Coliform Membrane Filter Procedure

Table 0-10. Microbial tests performed on bore hole soil samples.

The analyses included total heterotrophic bacteria (standard plate count) and total coliform bacteria. Tests were conducted according to Standard Methods (Greenberg et al., 1992). The materials used for the analyses were stored in sterile plastic bags and kept refrigerated until the tests were begun.

The membrane filter procedure was used for analysis of total heterotrophs and total coliform. Core extracts for determination of microbial activity were prepared by adding 250 mL of deionized water to approximately 50 g (wet weight) of soil. Six successive decimal (10x) dilutions were prepared from the mixture to produce ideal plate counts (total dilution factor of 10^{-7}). Total bacteria and coliform counts are reported as total colonies per gram of wet soil in Table 4.8.

RESULTS OF THE FIELD INVESTIGATIONS

1.11. Ground Water Monitoring

There were three objectives associated with the ground water monitoring program: 1) characterize the hydrochemical facies (general ground water characteristics) of each of the areas studied, 2) evaluate ground water environments specific to each site to develop an understanding of the contaminant transformation and transport processes, and 3) determine the impacts of on-site wastewater treatment and disposal systems on ground water quality. The results of these studies are discussed in the following sections.

1.11.1. General Water Quality Characteristics

One of the first tasks in analyzing the water quality data was to compare the parameters which have the greatest influence on the general chemistry of the ground water samples. This type of analysis is useful in the following ways: 1) it can assist in evaluating the quality of the analytical results, 2) analysis of major ions can give some indication of the origin of the water, 3) major water quality characteristics can be used to identify similarities and differences between ground water from different wells, and 4) the analysis is of significant value in determining the types of chemical and microbial transformations that may occur in a subsurface environment. Since nitrate is the only common constituent of domestic wastewater for which there is a ground water standard, its occurrence in the ground water monitoring program will be discussed separately in the next section.

Two general ground water characteristics are of particular interest in a ground water monitoring program, the major ion chemistry and the oxidation-reduction (redox) chemistry. Major ions are those which are present at milli-molar (mM or 10^{-3} M) concentrations and usually include sodium (Na^+), potassium (K^+), calcium (Ca^{2+}), magnesium (Mg^{2+}), chloride (Cl^-), sulfate (SO_4^{2-}), and bicarbonate (HCO_3^-). The major ion chemistry of a solution will affect many of the chemical and geochemical interactions that might be expected to occur in the water and will influence the behavior of most of the trace inorganic constituents. Some of the chemical phenomena that are affected by the major ion chemistry include: 1) the ionic strength of the

solution, 2) the hardness of the water, and 3) the solubility of important soil minerals, especially carbonates. Of special relevance to this study is that the ionic characteristics of ground waters usually have a unique ratio of the individual ions as a result of sampling location, source of recharge water, lithology and geochemistry of the aquifer, flow patterns in the aquifer, and solution kinetics (Fetter, 1980). Evaluation of the major ion composition allows direct comparison of water quality from multiple wells and can be used to provide evidence as to whether or not the wells produce water from the same geologic formation.

One of the best methods for summarizing and comparing the major ion chemistry of a series of water samples is to plot the major ion chemistry on trilinear diagrams, also known as Piper diagrams (Hem, 1985). Trilinear diagrams are prepared by calculating the percentage of the total cations and anions represented by the concentration of each of the individual ions in milli-equivalents/L (1 meq/L = 10^{-3} eq/L). Triangular plots are then prepared of these percentages for cations and for anions, then the information is combined into a single, four axis, diamond shaped plot. The results of each water analysis is represented as a single point on the diagrams. One variation of the trilinear diagram involves plotting each point using circles with each diameter proportional to ionic strength of the solution. The trilinear diagram is thus a pattern plot which allows visual comparison of the major ion composition and ionic strength for a numerous water samples on a single graph.

Trilinear diagrams of major ions in ground water for the Sandia Heights and East Mountain area results are given in Figure 0-1 and Figure 0-2 respectively. Water quality results of the Sandia Heights monitoring wells group in the center left of the upper diamond diagram (Figure 0-1). The tight grouping of the points demonstrates consistent water quality results between each sampling event. Furthermore, the plot demonstrates that water from the up gradient and the down gradient monitoring wells exhibit the same major ion chemistry which provides confirmation that they draw water from the same geological formation. Although it is possible that the up gradient well is drawing water from a perched aquifer, these results show that the water produced by both the up gradient and down gradient monitoring wells almost certainly are from the same aquifer. The low concentrations of alkalinity, calcium and magnesium is representative of the water coming off the west face of the mountains that has had little opportunity to equilibrate with the host rock (Hem, 1985).
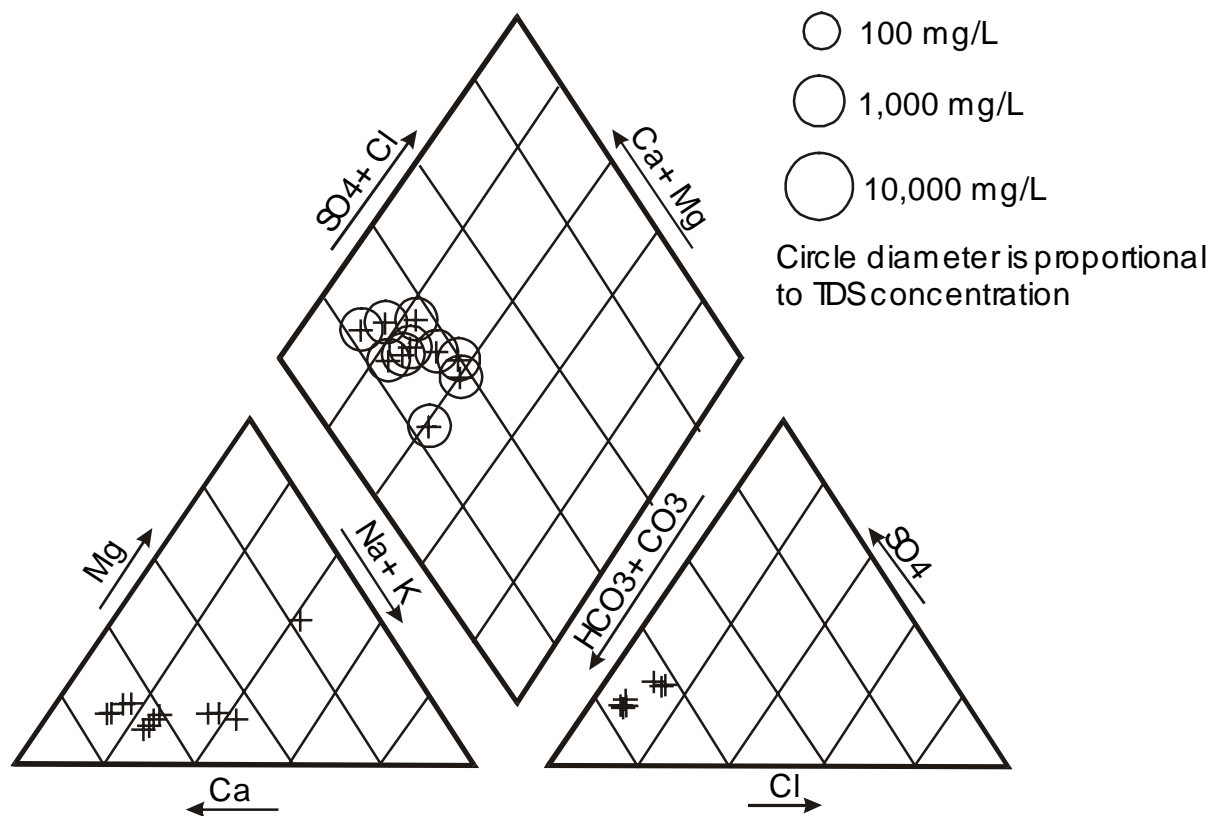
Figure 0-1. Trilinear plot of water quality data from the Sandia Heights monitoring wells.

In contrast to the consistent similarities in major ion water chemistry from the Sandia Heights monitoring wells, the East Mountain wells show two distinct groupings of water chemistries (Figure 0-2). The upper points are from the Piñon Ridge and Sandia Park wells and the lower points in the bottom corner of the diamond are from the Sierra Vista well. The dramatic difference of the Sierra Vista water analyses show a distinctly different chemical signature with the very high sodium levels and low fractions of calcium and magnesium. This may be the result of cation exchange with the silts and clays of the formation where the divalent Ca^{2+} and Mg^{2+} in the water are exchanging with the monovalent Na^+ and K^+ on the surface of the clays. This may indicate that the Sierra Vista well is completed in a different aquifer (possibly perched) or even a different formation (perhaps the Mesa Verde formation which is predominately fine grained sandstone with interbedded shale and coal). Another possible explanation for the high sodium concentration is the use of domestic ion exchange water softeners. The water quality in the East Mountain area is commonly very hard (i.e. high concentrations of calcium and magnesium). Ion exchange water softeners exchange Ca^{2+} and

Mg^{2+} for Na^+ on plastic resins. The resins are regenerated with a concentrated salt (NaCl) brine, which is sent to the on-site disposal system for disposal. Data collected to date does not permit confirmation of either hypothesis, although the Sierra Vista water quality does not have especially high chloride concentrations, which does not support the brine disposal theory.

Even though the results from the East Mountain monitoring wells plot separately on the diagram, the results for each well are very consistent between sampling rounds showing that (as in Sandia Heights) the general chemistry of each site remained constant throughout the duration of this study.
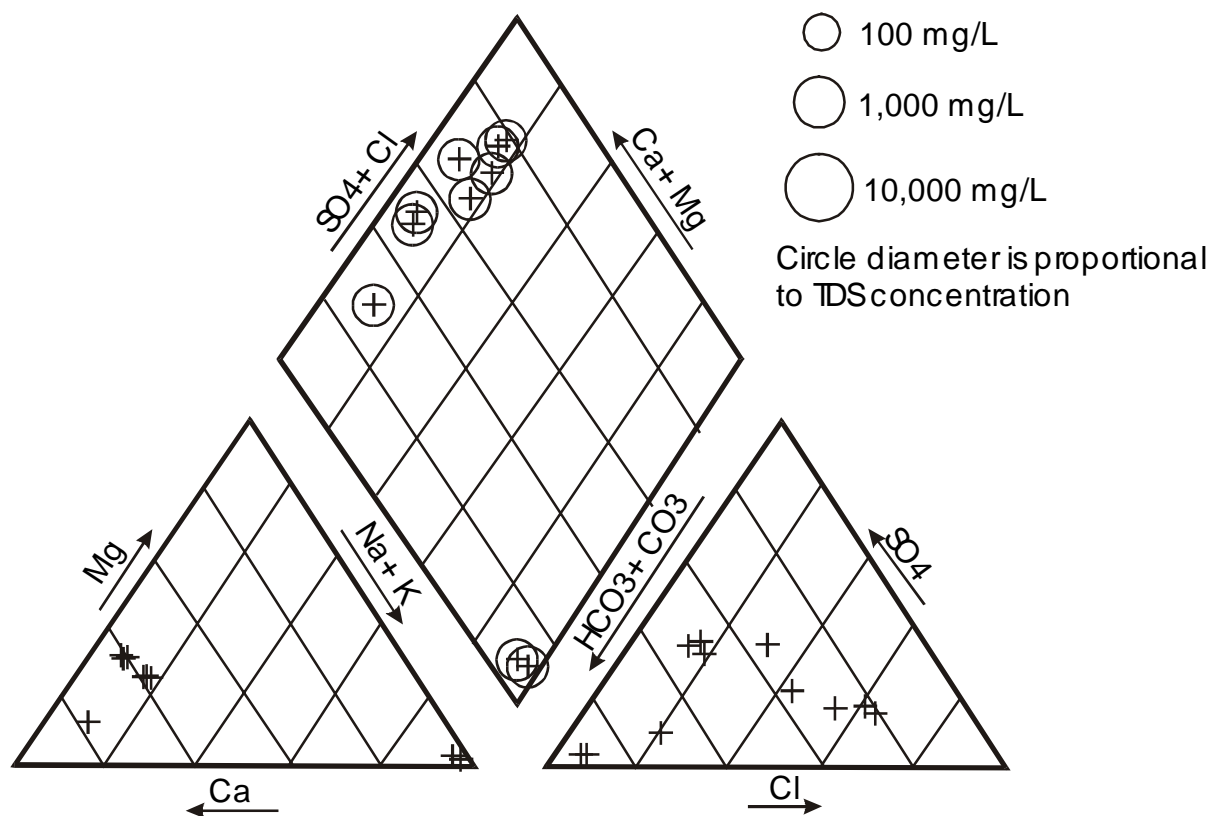
Figure 0-2. Trilinear plot of water quality data from the East Mountain area monitoring wells.

One of the most important factors associated with evaluating the fate of nitrogen species in soil and ground water environments is knowledge of the oxidation/reduction (redox) conditions in the subsurface environment. The discussion in Section 2.1 identified several important redox couples which are critical to the oxidation of reduced N species such as ammonia (NH_4^+), or conversely, the reduction of oxidized species such as nitrate (NO_3^-) and

nitrite (NO_2^-). There are several water quality parameters which can be used to indicate the redox conditions of a ground water sample. These include the concentration of dissolved oxygen (DO), total organic carbon (TOC), ammonium, soluble iron, and nitrates. Dissolved oxygen concentrations were measured in the field, while all other analyses were performed by the City of Albuquerque Water Quality Laboratory. The presence of DO and nitrates are indicators of oxidizing conditions as O_2 represents the highest energy electron acceptor normally available to microorganisms (Table 0-2), while nitrate is a close second. Both are rapidly consumed by heterotrophic organisms if an appropriate energy source is present such as TOC, or possibly by autotrophic organisms if ammonium is present. Degradable organic carbon will serve as an energy source for both aerobic and anaerobic microorganisms and is therefore considered a reducing agent in aqueous systems. The presence of ammonium suggests that there is insufficient O_2 for the nitrification reaction, therefore, the subsurface environment can be assumed to be anoxic. Iron plays an important role in interpreting the geochemistry of the suibsurface environment. This is based on the fact that iron(III) is very insoluble above pH 3, forming a $Fe(OH)_{3(s)}$ precipitate, while iron(II) is generally soluble. Furthermore, iron(III) minerals can serve as a terminal electron acceptor for oxidation of carbonaceous substrates thereby producing soluble iron(II) (Table 0-2). Thus, the presence of soluble iron is indication of reducing conditions in the formation that are likely due to biological activity.

It is possible to measure a field redox potential (Eh) using a platinum electrode and a high impedance volt meter (i.e. a pH meter) (Hem, 1985). However, work by Lindberg and Runnels (1989) has shown there is almost always little or no correlation between field measurement of this parameter and reaction couples which control the redox chemistry of the solution, therefore Eh was not measured in this study.

Table 0-1 and Table 0-2 summarize the water quality data collected from four quarters of ground water sampling and include the concentrations of parameters which are indicative of the redox conditions for ground water samples from the Sandia Heights and the East Mountain monitoring wells. The numeric values in these tables represent average concentrations over all samples taken during the course of this study. (There was little variability in the constituent concentrations throughout this study as illustrated by the nitrate values plotted in Figure 0-3 and Figure 0-4. All data are tabulated in Appendix 6) All of the Sandia Heights wells show evidence

of oxidizing conditions, as well as moderately high pH and alkalinity. The DO values are all greater than 4 mg/L which is well above the value of 1 mg/L below which the growth of aerobic organisms becomes limited by lack of dissolved oxygen. Nitrate concentrations are greater than 1 mg N/L, while TKN and ammonium concentrations are below detection limits. These results are consistent with the interpretation that nitrification has oxidized all of the reduced nitrogen (i.e. organic N and ammonium) to nitrate.

The low concentrations of TOC suggest that there is an inadequate supply of organic carbon to support denitrification of all of the nitrate present in the down gradient wells. The exact stoichiometry of the microbial denitrification reaction depends on the chemical form of organic carbon source with reduced forms of carbon such as methane (CH_4) capable of reducing more nitrate than more oxidized compounds such as formic acid ($HCOOH$). Assuming the TOC in the ground water is represented by the generic formula CH_2O, the stoichiometry of the denitrification reaction (equation 2.10) shows that five moles of organic carbon represented as CH_2O is needed to reduce four moles of nitrate, which corresponds to1.1 mg/L of carbon to reduce 1 mg N/L as nitrate. A further point to note is that the ability to detect TOC in ground water samples in the presence of such high DO concentrations indicates that these compounds may not be readily biologically degradable. Therefore, these compounds would not be a suitable carbon source for the denitrification reaction even if anoxic conditions were present.

No coliform bacteria were detected in the Sandia Heights monitoring wells. Based on the large depth to ground water this is not a surprising result as well graded soil is an effective filter for removal of bacteria sized particles.

Parameter (mg/L)	EG01 (Elena Gallegos)	SH02 (Cedar Hill)	SH01 (Mhoon)	SH03 (San Rafael)
TKN	<5	<5	<5	<5
NH_3	<.01	<.01	<.01	<.01
D.O.	4.7*	5.5*	7.2*	not measured
TOC	1.41	0.454	.355	.770
NO_3^-	0.097	3.49	1.36	1.41
Cl^-	7.45	17.1	6.77	4.94
Fe (Total)	.008	1.17	1.90	0.012
SO_4^{2-}	49.3	43.5	27.1	32.6
coli bacterium (1=present, 0= not present)	0	0	0	0
MBAS	0	.02	0	0

*-Average of 3 D.O. measurements

Table 0-1. Summary of water quality analyses for Sandia Heights monitoring wells (average of four sample events).

In contrast to the Sandia Heights ground water samples, ground water samples collected from the Piñon Ridge and Sierra Vista monitoring wells suggest that strongly reducing conditions are present. Furthermore, these results provide some evidence of direct contamination of the ground water by on-site disposal systems. The low DO levels and absence of nitrates in the Piñon Ridge and Sierra Vista wells are both indicative of reducing conditions which prevent the nitrification reaction. Supporting evidence includes the detectable concentrations of ammonium and iron, and the low sulfate concentrations. The TOC concentrations in all of the East Mountain water samples were substantially higher than in the Sandia Heights wells. Based on the simple stoichiometry presented in equation (2.10), there appears to be sufficient concentrations of organic carbon to support microbial denitrification, which may constitute a nitrate removal mechanism. The Sierra Vista well also had a faint odor of sulfides from the water which suggests that sulfate reduction is occurring. Microbial sulfate reduction requires anaerobic conditions which are also required for denitrification. Both the Mancos Shale and the Mesaverde Formation contain coal seams (Kelley and Northrop, 1975). Locally, there is a coal seam approximately one mile to the northeast of the Sierra Vista subdivision. A coal seam with its

attendant high concentrations of organic compounds would be expected to provide strongly reducing conditions in associated ground water. For example coal gas has high concentrations of methane (CH_4) which is an excellent substrate for many aerobic and anaerobic organisms, including sulfate and nitrate reducers. These coal formations may also be the cause of the high TDS concentrations as ground water accumulates dissolved minerals as it passes through the marine shales of the Mancos. The Sandia Park wells (SP01 and SP02) have higher DO and nitrate concentration indicating oxidizing conditions. The presence of confined groundwater in Sandia Park well number 1 (SP01) was unexpected and may partly explain the low nitrate concentrations (3.48 mg N/L) detected. This well is located approximately 200 m from a domestic well that has been regularly sampled by the USGS (Rankin, 1995) and has nitrate concentrations averaging about 20 mg N/L. The presence of a confining layer will limit transport of contaminants vertically, thus, if the contaminants are due to on-site disposal systems, deeper formations would have lower concentrations of the pollutants.

The value of 0.33 for Coliform bacteria reflects detection of these organisms in one of three samples taken.

Parameter (mg/L)	SV01 (Sierra Vista)	PR 01 (Piñon Ridge)	SP01 (Sandia Park 01)	SP02[1] (Sandia Park 02)
TKN	0.3	0.43	<5	<5
NH_3	0.11	.02	< 0.01	< 0.01
D.O.	1.3	0.8	$5.3^{[2]}$	-
TOC	2.63	4.57	3.77	1.85
NO_3^-	<0.05	0.37	3.48	0.60
Cl^-	31.7	83.2	268	45
Fe (Total)	.604	.295	.006	<.005
SO_4^{2-}	<7	209	103	27.4
coli bacterium[3] (1=present, 0= not present)	0	0.33	0	0
MBAS	0	0.05	0	0

[1] = A single sample of well SP02 was taken
[2] = The D.O. values for well SP01 represent the average of two readings
[3] = Wells SV01, PR01 and SP01 were sampled 3 times for coliform bacteria.

Table 0-2. Summary of water quality analyses for East Mountain area monitoring wells (average of four sample events).

1.11.2. Nitrate Concentrations

Reference to *-Average of 3 D.O. measurements

Table 0-1 and Table 0-2 shows that the nitrate concentrations in water samples from the monitoring wells are well below the State ground water standard of 10 mg N/L as NO_3^-. Figure 0-3 and Figure 0-4 are plots of the nitrate concentrations for each monitoring well as a function of time. Note that the second Sandia Park well (SP02) only had one ground water sample taken and therefore is not included in these plots. These plots show that the nitrate concentrations did not vary seasonally over the duration of this study.

There are three possible phenomena that might explain low nitrate concentrations in ground water beneath a residential area with a high density of conventional wastewater treatment and disposal systems:

- A nitrate plume is developing and concentrations are increasing but have not yet reached their maximum values.

- Anaerobic conditions prevail in the subsurface environment, therefore nitrification is inhibited, and a nitrate plume never fully develops.

- Aerobic conditions prevail in the unsaturated zone resulting in nitrification, however, denitrification in deeper strata reduce nitrates.
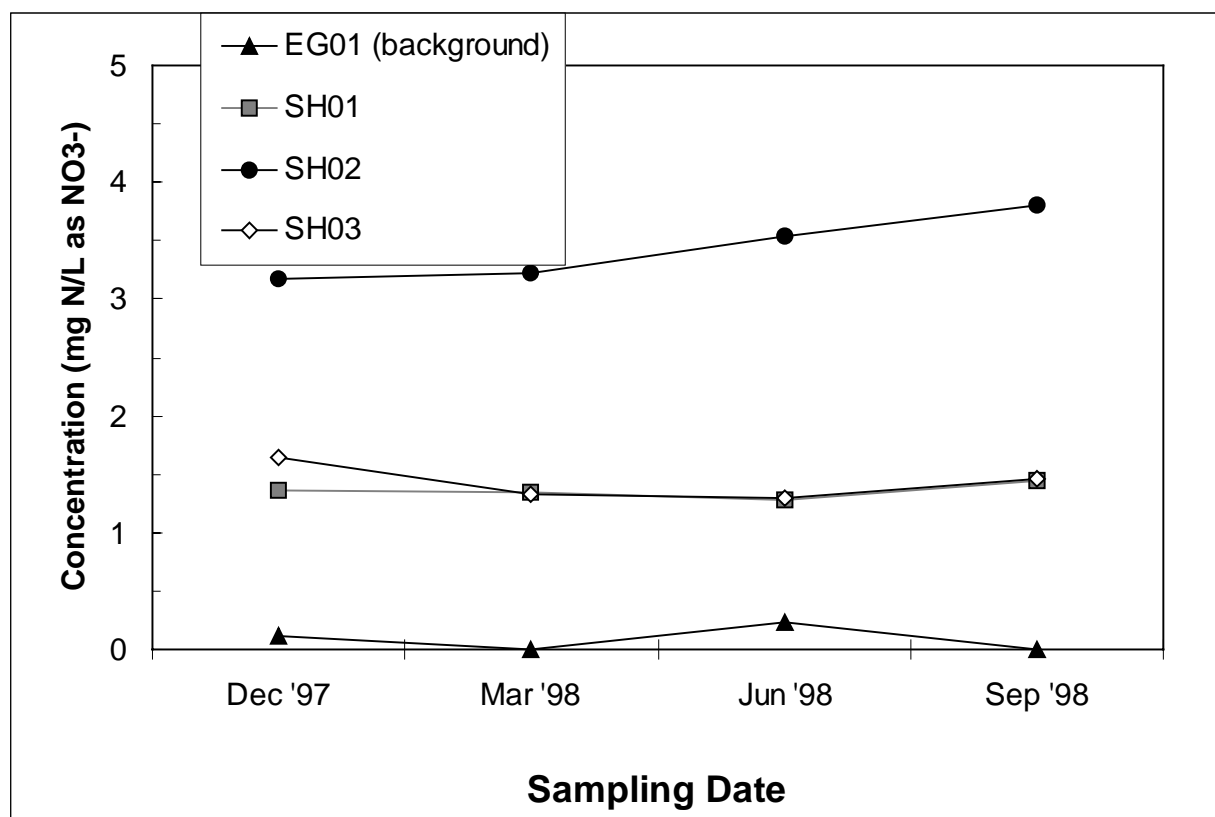
Figure 0-3. Nitrate concentrations versus time for the Sandia Heights monitoring wells.

Careful examination of the plots in Figure 0-3 suggest there may be a trend of increasing concentrations in the Sandia Heights monitoring wells. Due to the small value and the limited number of data points, it is not possible to confirm an increase statistically, but there appears to be an upward trend in all three of the down gradient wells. This is especially relevant in light of the ground water modeling results which also suggest that nitrate concentrations at the down gradient boundary of the subdivision may be increasing (discussed in Section 5). As discussed in Section 4.1.1, the ground water chemistry in this formation indicates the presence of oxidizing conditions which are conducive to production and continued stability of nitrates. An upward trend in nitrate concentrations is consistent with this observation.
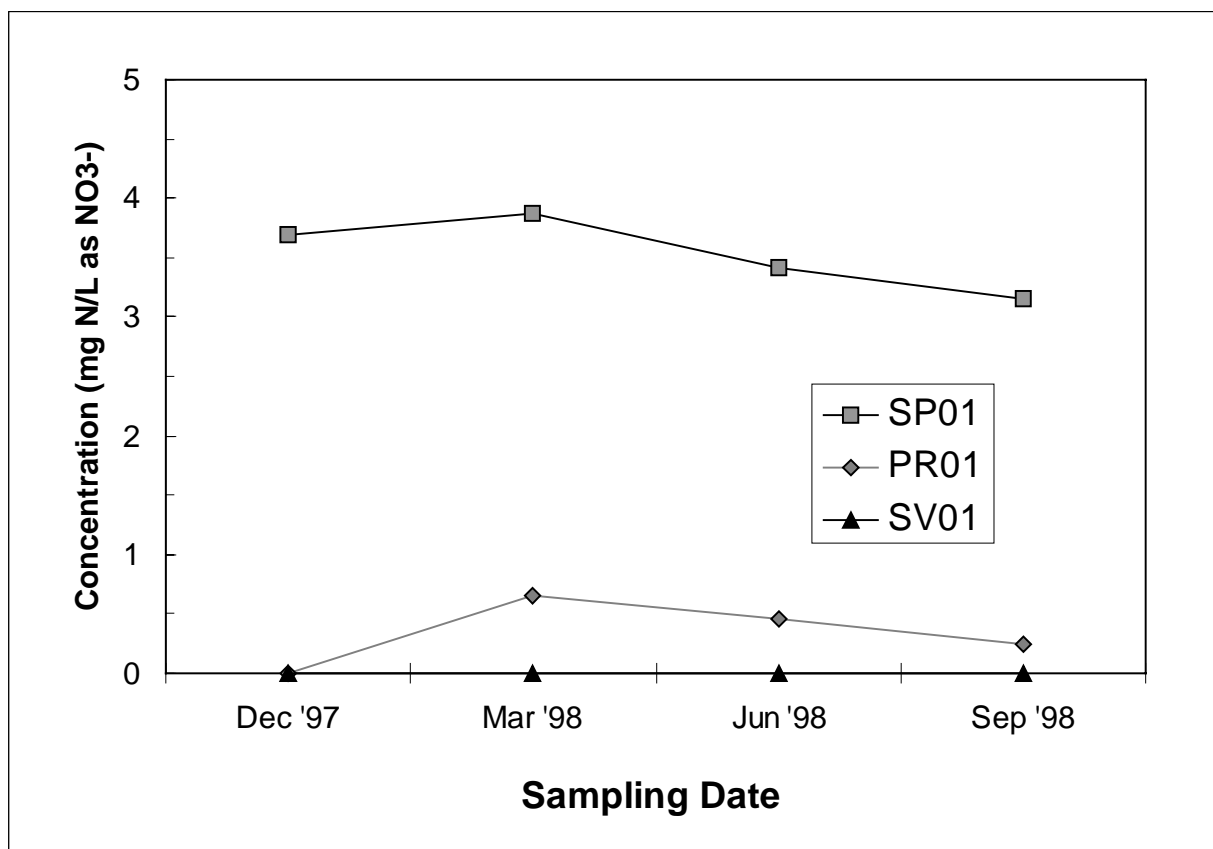
Figure 0-4. Nitrate concentrations versus time for the East Mountain monitoring wells.

In contrast to the data from the Sandia Heights subdivision, the data for the East Mountain ground water samples suggests that nitrate concentrations may be decreasing with time. This trend, if in fact it is occurring, would be consistent with the observation that most of the indicators of the redox conditions in the East Mountain area monitoring wells suggest that conditions appropriate for microbial denitrification may be present. The presence of reducing conditions in the subsurface environment is believed to be in part responsible for the low nitrate concentrations in the East Mountain monitoring wells.

Detection of long term trends based on only four data points is problematic at best. Kues and Garcia (1995) plot three years of monthly monitoring data for 20 East Mountain domestic wells which shows a high degree of variability in many of the analytes. However, no statistical analysis of their data is presented, nor are the data tabulated which might allow this type of analysis. Visual inspection of nitrite and nitrate plots for six of the East Mountain wells near the three areas considered in this investigation show apparent decreasing levels of nitrite plus nitrate

in three of the wells (well numbers 11, 16 and 20 in the Kues and Garcia (1995) report), increasing concentration in one well (well number 2), no change in one well (well number 15) and a long term decrease followed by an increase (well number 10). Perhaps the most significant conclusion that can be made of this data is that long term trends will be difficult to identify without long periods of record. This is therefore an argument in favor of continued sampling and analysis of the monitoring wells constructed in this study.

During the course of this investigation it was suggested that natural organic matter (NOM) in some of the East Mountain geologic formations, particularly the Mancos Shale, might be a suitable substrate to support growth of denitrifying organisms. Accordingly, a brief microcosm study was conducted to determine if denitrification could be demonstrated in a mineral salts media containing soil and a microbial inoculum. This study was done by an undergraduate student during the summer of 1998 and is described fully in Appendix 8. A brief discussion is provided here.

A sample of Mancos shale was obtained from a depth of about two feet below the surface from an outcropping near State Highway 14 near Cedar Crest. By virtue of being located near the surface this sample was highly weathered and probably isn't entirely representative of shales from greater depths. Samples of this soil were ashed at 550 C which produced a 3.24% loss of mass upon ignition indicating a low organic content. 50 g of shale was leached with deionized water which was then analyzed for COD and ammonia. There was no detectable ammonia in the leachate and the COD of the shale was 0.40 mg COD/g shale.

The microcosm study consisted of a two week investigation. Six erlenmeyer flasks were prepared, all filled with 100 mL of a tap water solution containing 50 mg N/L as NO_3^-. Three of the flasks received 25 g of Mancos shale and three received 25 g of acid washed silica sand which served as a control. 1.0 mL of wastewater effluent was added to each flask to serve as a microbial seed culture, the flasks were purged with $N_{2(g)}$ to remove oxygen and then were incubated at 20 C. 4.0 mL samples were collected from the flasks periodically, after which each sample was again purged with $N_{2(g)}$. The results of the microcosm study are presented in Figure 0-5. This study shows no discernable difference in the nitrate concentration between the microcosms containing the Mancos shale and those containing acid washed sand, thus it is apparent that denitrification did not occur over the duration of this experiment. While the results

of this brief study are not definitive, they do indicated that the shale has low concentrations of organic carbon and that it will not serve as a substrate for rapid denitrification.
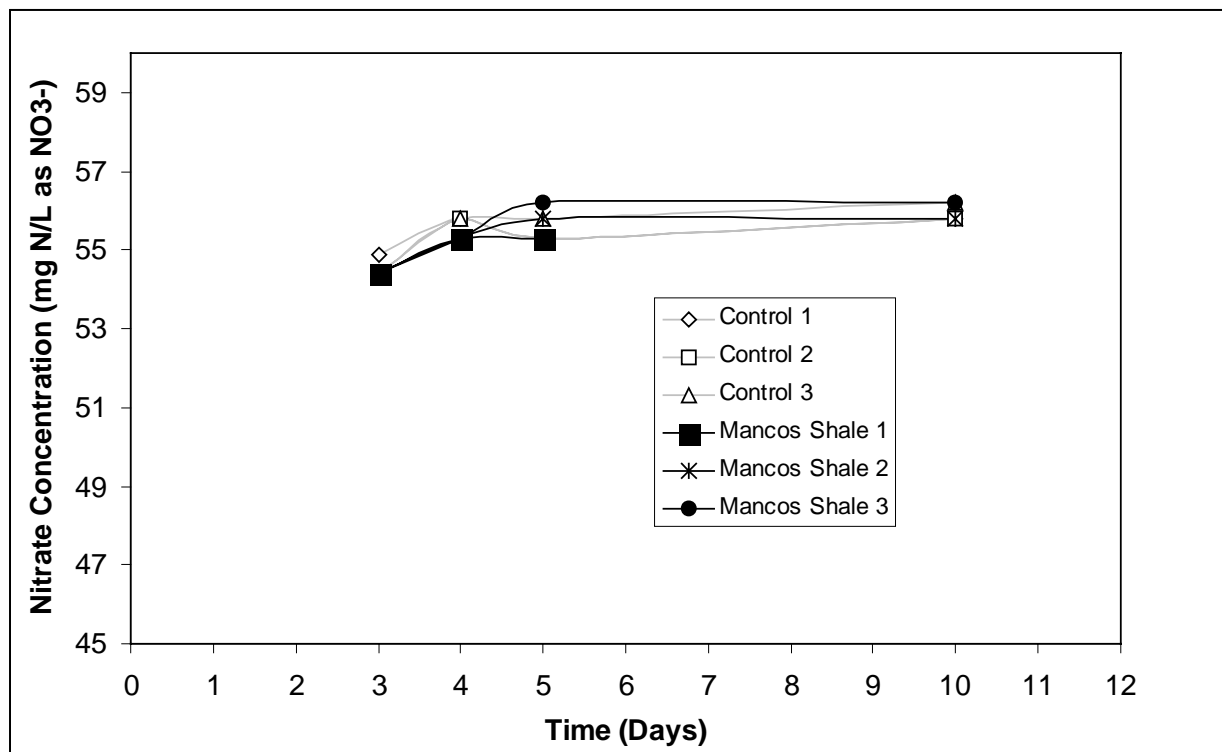
Figure 0-5. Nitrate concentration versus time for the microcosm study.

1.11.3. Other Indicators of Ground Water Quality

In addition to major ion chemistry and redox couples, several other constituents of domestic wastewater can serve as indicators of ground water contamination from on-site disposal systems. This includes chlorides, coliform bacteria, methylene blue active substances (MBAS) and synthetic organic chemicals (SOCs) such as solvents, pesticides, herbicides and other compounds which are often introduced to the disposal system through household use (Bashar et al., 1990, Reneau et al., 1989, Tuthill, et al., 1998)

Chlorides, bacteria, and household chemicals have been used as indicators of septic system impact to ground water by a number of investigators (Alhajjar, et al., 1990, Tuthill, et al., 1998, Reneau, et al, 1989). The average concentrations of chlorides, as well as the presence of bacterium and other indicators of impact from wastewater disposal systems are shown in Table

0-1 and Table 0-2, for Sandia Heights and the East Mountain monitoring wells. Chloride, in particular, has value as a tracer because it is non-reactive and readily moves through most soil and groundwater systems. Values of chloride concentrations in wastewater depend on the lifestyle and activities of the residents, however, there is little data available on the increases due to domestic use. Brandes (1978) presented a table which shows that chloride concentrations increase on average by 46 mg/L in grey water septic tanks and 96 mgL in black water septic tanks, however, this data set appears to be limited to two households. Metcalf and Eddy (1991) present a table which shows the average chloride concentration of municipal wastewater as 50 mg/L with a range from 30 mg/L to 100 mg/L, but it is not clear what the influent chloride concentration might be. Sawyer et al. (1994) claim that household use increases the chloride concentration of municipal wastewater by about 15 mg/L. As discussed previously, a further complication is that many of the households in the East Mountain area have ion exchange water softeners which are regenerated with concentrated salt (NaCl) brine. Disposal of this brine in absorption field wastewater disposal systems will greatly increase the sodium (Na^+) and chloride (Cl^-) concentrations of the underlying ground water. Thus, while it is clear that the chloride concentration will be elevated in the absorption field leachate, it is not known what the actual value of this increase will be.

A plot of chloride concentration versus nitrate concentration is presented in Figure 0-6 and **Error! Reference source not found.**. Evidence of contamination from on-site wastewater disposal systems would be expected to appear as a correlation between these two variables, however, no such relationship is apparent from the data collected. There are a couple of possible explanations. First, natural variability of chloride concentrations in the ground water may overwhelm any correlation due to wastewater characteristics. For example, the chloride concentration in the Sandia Park monitoring well samples averages 268 mg/L compared to less than 10 mg/L in the Sandia Heights wells. Thus, an increase in the chloride concentration of a few tens of mg/L might difficult to detect. A second factor which may prevent a correlation between chlorides and nitrates is that water quality in the East Mountain area wells is consistent with the presence of reducing conditions in most of the wells. It is likely that no correlation between chloride and nitrate will exist if nitrate cannot be generated due to lack of oxygen.

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Reference to Figure 0-6 suggests there may be a weak correlation between chloride and nitrate concentrations in the Sandia Heights ground water samples, however, with only four points it is not possible to confirm this correlation. For the East Mountain area water samples (**Error! Reference source not found.**) the apparent correlation between the two parameters is misleading in that only one monitoring well (Sandia Park 01) had significant nitrate concentrations, although the average nitrate concentration in this well was still well below 10 mg N/L as NO_3^-. The very low nitrate concentrations in the other three wells results in the points representing these values being clustered near the origin.
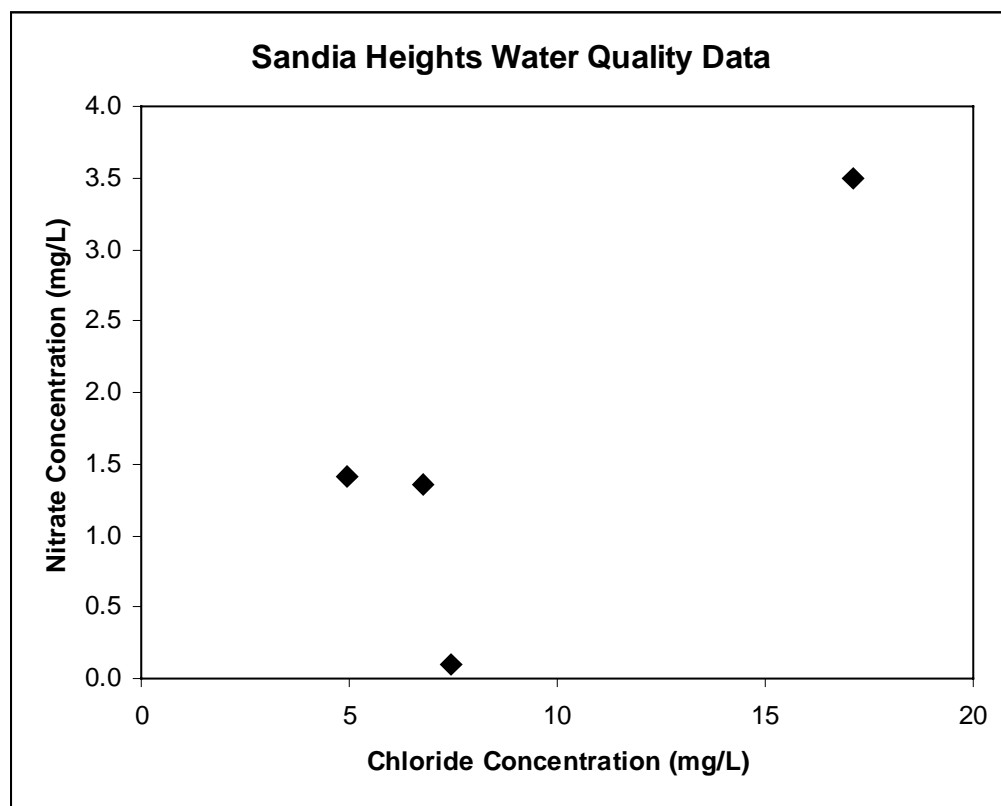
Figure 0-6. Plot of average chloride concentration versus average nitrate concentrations for Sandia Heights water samples.
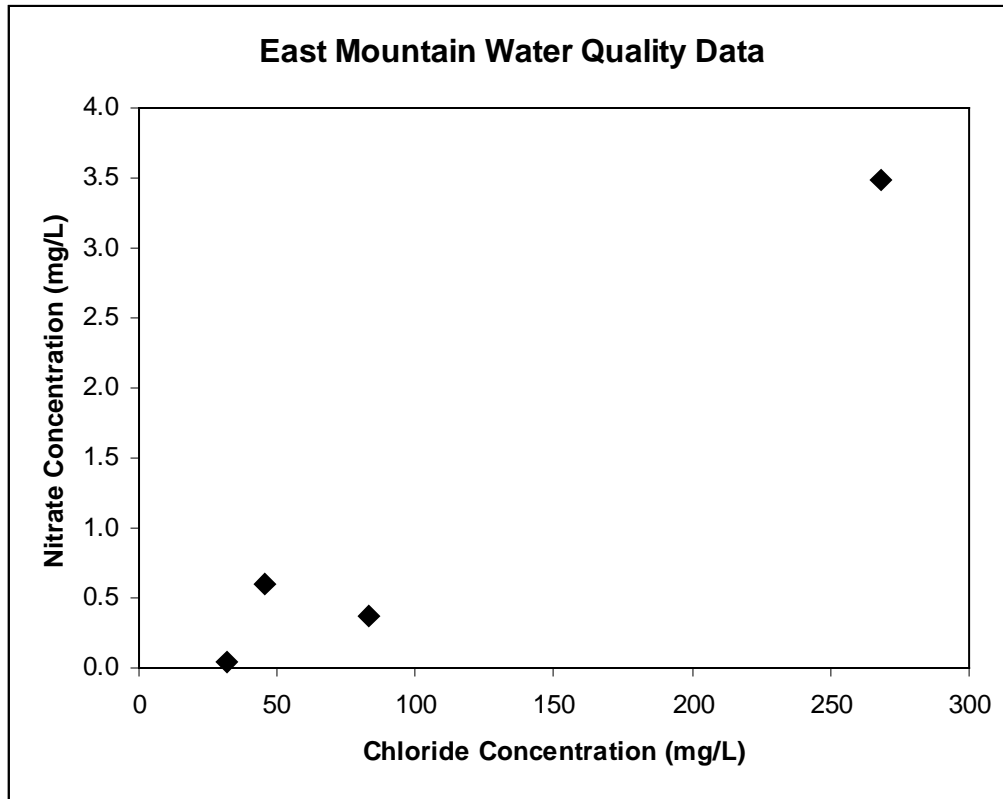
Figure 0-7. Plot of average chloride concentration versus average nitrate concentrations for East Mountain area water samples.

The presence of coliform bacteria in a ground water sample is a strong indication of contamination of the water by domestic wastewater. Laboratory analyses detected coliform bacteria in half of the Pinon Ridge samples. Initial samples containing coliform bacteria was suspected of being a result of contamination of the well during construction or subsequent sampling operations, so all of the wells were dosed with liquid sodium hypochlorite (NaOCl - bleach) and additional sample equipment disinfection procedures were implemented. Positive tests for the presence of coliform bacteria were subsequently found in one of three samples, thus suggesting contamination of the ground water from domestic wastewater. Conventional wisdom states that just a few meters of well graded soil will remove bacterial particles from septic tank leachate through physical filtration and predation by soil microorganisms (Reneau et al., 1989). Therefore, the presence of coliform bacteria in a well of this depth might be an indication of direct contamination of the ground water through a fracture flow process which has been shown to be an important mechanism for ground water flow in this area. However, the inconsistent

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results and the lingering possibility that bacterial contamination of the well was the result of construction activities or the sampling process does not allow this conclusion to be drawn with a high degree of certainty.

Methylene Blue Active Substances (MBAS) are compounds which transfer a blue color (methylene blue) to an immiscible organic liquid when mixed with water. MBAS compounds are principally anionic surfactants which are contained in some detergents, therefore, their presence is an indicator of contamination by domestic wastewater. They are not subject to any federal or state water regulations, but were analyzed as an indicator of ground water contamination. MBAS was measured at 0.3 mg/L in one sample from the Piñon Ridge well. Although this result is not definitive, as with the coliform tests it does suggest the possibility of contamination from on-site wastewater disposal systems.

The three East Mountain area wells (all except SP02) and two of the Sandia Heights wells were sampled and analyzed by the USGS for 68 organic compounds which are frequently associated with household solvents, foods, and other products. A complete listing of the analytes is presented in Table 0-6 and the list of compounds actually detected is presented in Table 0-3.

	Concentration (ug/L)			
Compound	EG01	SV01	PR01	SP01
2,6-di-t-p-benzoquinone		0.13	0.37	
tri (2-chloroethyl) phosphate			0.073	
NPEO1 - total			1.3	
fluoranthene			0.029	
pyrene			0.072	
bis(2-ethylhexyl) phthalate			4.2	
1,2,4-trimethylbenzene		0.22		0.22
D-limonene		0.066		
naphthelene		0.43		
naphthelene, 2-methyl		0.54		0.081
naphthelene, 1-methyl		0.38		
2(2-btxyethxyl) ethyl acetate				0.16
tribromomethane	7.6			0.64

Table 0-3. Trace organic compounds detected in ground water samples by the USGS.

First, note that all of these compounds were detected at very low concentrations and the possibility exists that these values may simply represent contamination of the water samples during sample collection, transport, or analysis. Probably the most common and general indicators of waste water contamination detected by this method are the nonylphenol ethoxylates (NPEO1 and NPEO2) which are composed of a total of about 20 isomeric compounds which are directly associated with nonionic detergent use. They were detected in the Pinon Ridge well.

The results of these organic analyses should be interpreted with caution as it is possible that the organics are a result of contamination during well construction. Considerable effort was made to limit potential contamination by steam cleaning all equipment prior to drilling and through decontamination of all sampling equipment after sampling each well. Furthermore, most of the wells were purged and sampled at least twice as part of the regular sampling program before they were sampled for these organic constituents so that contamination resulting from well construction should be negligible. Table 0-4 lists the interval between the date each well was constructed and when it was sampled by the USGS for organic constituents.

Well	Date Drilled	Dated Sampled by USGS	Age of Well on Day of Sampling (mos.)
EG01	-	7/1/98	-
SH03	8/26/98	7/1/98	10.3
SV01	9/15/97	6/17/98	9.2
PR01	9/9/97	6/18/98	9.4
SP01	9/22/97	6/18/98	9.0
SP02	10/12/98	12/3/98	1.7

Table 0-4. Time between well construction and sampling for organic constituents.

The rate of transport of these compounds through the subsurface environment and the potential for their biotic or abiotic degradation is not known, thus, it is not possible to even qualitatively estimate the contribution that on-site disposal systems make to the ground water near the top of the water table. Nevertheless, the fact that these compounds are present provides evidence that on-site disposal systems may have impacted the ground water beneath these subdivisions. It is clear that further sampling of these wells for organic contaminants is needed.

1.12.   Pollutant Transport and Transformations in the Vadose Zone

The objective of vadose zone field study was to determine the transformation and transport processes occurring in the subsurface soils beneath an absorption field. This investigation consisted of drilling five holes to a depth of 65 ft below ground level and collecting core samples at five foot intervals. A summary of the methods used in this field study are described in Section 3.2. The five holes were designated A through E. Holes A and B were background holes, while holes C through E were drilled immediately adjacent to the leach field. Hole C was nearest the beginning of the absorption field and is believed to receive a heavier loading of the septic tank effluent than soils in the other bore holes. A map of the site is presented in Figure 0-12.

1.12.1. Soil Properties Test Results

Soil samples taken from hole C were subjected to a range of standard geotechnical tests in order to obtain basic properties and description of the soils beneath the leach field. These included measurement of the soil's grain size distribution, Atterberg limits, and unsaturated hydraulic characteristics. The results of these tests are described below.

The grain size distribution of soil samples retrieved from hole C are presented in Appendix 7. These distributions reveal that the soils are comprised principally of sand-sized particles. The grain size distributions are used to develop the soil classifications

The soil samples retrieved from hole C were classified according to the Unified Soil Classification System (USCS) method (ASTM 2487). This method utilizes the grain size distribution and plasticity parameters where appropriate. The classification results are given in Table 0-5.

All of the soils in bore hole C classify as coarse-grained soils which is consistent with the visual observation that the soils are predominantly "sandy." The classification also show that the soil profile beneath the leach field is not a single, uniform soil. Rather, the soils exhibit considerable variability with depth. Observations during drilling suggest that the soil is variable laterally as well. In other words, at a given depth the soil texture can vary from hole to hole.

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The saturated hydraulic conductivity and the moisture characteristic curves were measured on soil samples from hole C. The moisture characteristic curve data was fit to the commonly used van Genuchten function (1980) (equation 3-5). These results are summarized inTable 0-5.

Depth (ft)	SCS classification Group symbol	Soil description	K_{unsat} (cm/sec)	$\theta_s{}^1$	$\theta_r{}^2$	n^3	α^3 (cm^{-1})
15	SM	Silty sand	2.0×10^{-3}	.25	.09	2.24	.082
20	SW-SM	Well-graded sand with silt	5.8×10^{-3}	.30	.08	3.24	.101
25	SW-SM	Well-graded sand with silt	2.4×10^{-1}	.29	.01	1.45	.862
30	SP	Poorly-graded sand	2.3×10^{-1}	.32	.08	2.27	.131
35	SC	Clayey sand	1.3×10^{-3}	.40	.12	1.50	.171
40	SC-SM	Silty, clayey sand	8.4×10^{-4}	.43	.17	1.63	.150
45	SC	Clayey sand	1.3×10^{-3}	.38	.15	1.62	.229
50	SM	Silty sand	4.9×10^{-3}	.30	.15	1.81	.082
55	SW-SM	Well-graded sand with silt	2.4×10^{-1}	.34	.08	2.29	.156
60	SM	Silty sand	2.9×10^{-3}	.43	.16	1.29	.207
65	SC-SM	Silty, clayey sand	8.7×10^{-3}	.45	.20	1.63	.343.

[1]Volumetric moisture content under saturated conditions
[2]Residual moisture content
[3]Empirical constants to fit the van Genuchten equation.

Table 0-5. Unsaturated hydraulic properties of soils from bore hole C as a function of depth.

The soil samples from the intervals that have no fine soil component (classified as SW or SP) have saturated hydraulic conductivities in the range typical of clean, coarse sands. The remainder of the soils possess hydraulic conductivities consistent with silty sands.   The van Genuchten parameters are in the ranges expected for sands of various textures for a wetting path. These parameters depend upon the history of the specimen, i.e. wetting and drying path. For

example, the values of θ_s are not always coincident with the porosity because air trapped in the interstitial voids during wetting reduce the water content.

The measured gravimetric moisture content for each soil sample is presented in Table 0-6, expressed as a percent and is plotted in Figure 0-8 and Figure 0-9.

Depth (ft)	Hole A	Hole B	Hole C	Hole D	Hole E
6	2.23	10.65	5.98	-	-
8	3.13	4.14	-	-	-
10	3.29	3.78	5.01	-	-
15	5.68	8.11	9.54	5.80	4.29
20	6.39	6.89	5.88	21.84	2.57
25	7.36	7.41	6.26	5.20	3.88
30	7.59	4.74	4.20	7.77	2.80
35	9.37	2.88	11.44	12.53	2.44
40	1.65	7.07	12.22	7.95	1.78
45	5.77	23.19	14.55	5.33	5.27
50	6.33	7.97	11.13	3.48	5.63
55	2.45	5.09	9.73	6.49	2.31
60	4.02	11.84	14.83	4.33	5.05
65	13.27	4.27	12.58	7.33	8.06

Table 0-6. Gravimetric moisture content (%) for soils from vadose zone sampling program.

The gravimetric moisture content in soil samples from the background holes vary from about 2 to 23%, but 86% of the samples are in the range of 2% to 10%. The value of 23.2% measured from a soil sample collected at 45 ft depth from Hole B is believed to be high due to laboratory error. These results are consistent with variability of the soil profile expected from the soil classification data and observations during drilling. The results indicate the presence of elevated water content in hole C, which is near the selector valve. The two outlying values in holes B and D may be due to local clay lenses at these depths. These fine-grained soils will retain much greater amounts of water compared to the sandy soils. The measured water contents suggest that the majority of the septic tank effluent was discharged (at least recently) in the vicinity of hole C and that effluent does not infiltrate into the soil uniformly along the distribution length of the pipe.

The water content from soil samples in hole C suggest that elevated water contents are only present below 30 feet. This result most likely indicates that there is little horizontal spreading of water as it infiltrates, at least in the coarse, loosely consolidated near-surface soils. The bore hole is believed to have been located five feet from the leach line, however, its exact location was not known with precision. It is also possible that these results are a consequence of water being infiltrated into the soil intermittently and not at a constant rate. In this scenario, the lower water contents below 30 feet may reflect a pulse of effluent moving downward past this depth at this point in time.

1.12.2. Discussion of Physical Characteristics of Subsurface Soils

The soils underlying the leach field are generally coarse-grained soils, with relatively large hydraulic conductivities. As the effluent from the leach field infiltrates into the soil, it will move downward relatively quickly, and only relatively small amounts of water will be retained by the soil. One estimate of the amount of water a soil retains in response to infiltration is the field capacity, which is the water content remaining in the soil after free or gravity drainage has occurred. Field capacity is often arbitrarily reported as the water content of the soil at about 3500 mm of suction head (Jury et al., 1991). Below field capacity, the hydraulic conductivity is often assumed to be so low that gravity drainage becomes negligible and the soil moisture is held in place by suction. It is important to recognize the approximate nature of the estimate of storage capacity based on the field capacity concept. The common definition of field capacity used here is arbitrary, and ignores other factors that affect the amount of water retained in a soil layer. However, the field capacity concept is widely used because it fills the need for a simple parameter that defines the storage capacity of a soil layer. In this report, field capacity is used for comparative purposes to provide a first order estimate of the water storage capacity of a soil layer independent of the underlying waste or foundation soil properties.

From the parameters in Table 0-5, the field capacity (based on gravimetric drainage of the soil) is estimated to be on the order of 3 to 13%, with an average of 8%. These values reveal how little water these coarse soils will retain and suggest that water will move rapidly downward through the soil horizon. These estimates of field capacity also suggest that the water contents in

the lower portions of hole C may be due to a transient "pulse" of effluent that is moving downward.

Figure 0-8 is a plot of the gravimetric moisture content for each of the five bore holes with depth (data from Table 0-6), while Figure 0-9 plots the data for holes A (background) and C (closest to absorption field) to facilitate direct comparison. The information contained in this figure is critical because it is used to calculate the concentrations of the chemical constituents in the soil pore water. There are a couple of points that should be noted. First, there is a high degree of variability in the soil water content. This is due to natural variability of the soil particle size; fine grained soils retain more water than coarse grained soils. Second, the water content in most of the samples is very low, typically less than 10%. Again, this is due to the coarse nature of the soils which reduces their ability to retain water. Finally, the water content in the background holes (A and B) is generally lower than in the hole C which is closest to the beginning of the absorption field. This phenomena is the result of continuous application of water to the leach field which subsequently percolates to depth. The high water content near the surface is also partly explained by moisture removal from the background soils by evapotranspiration.
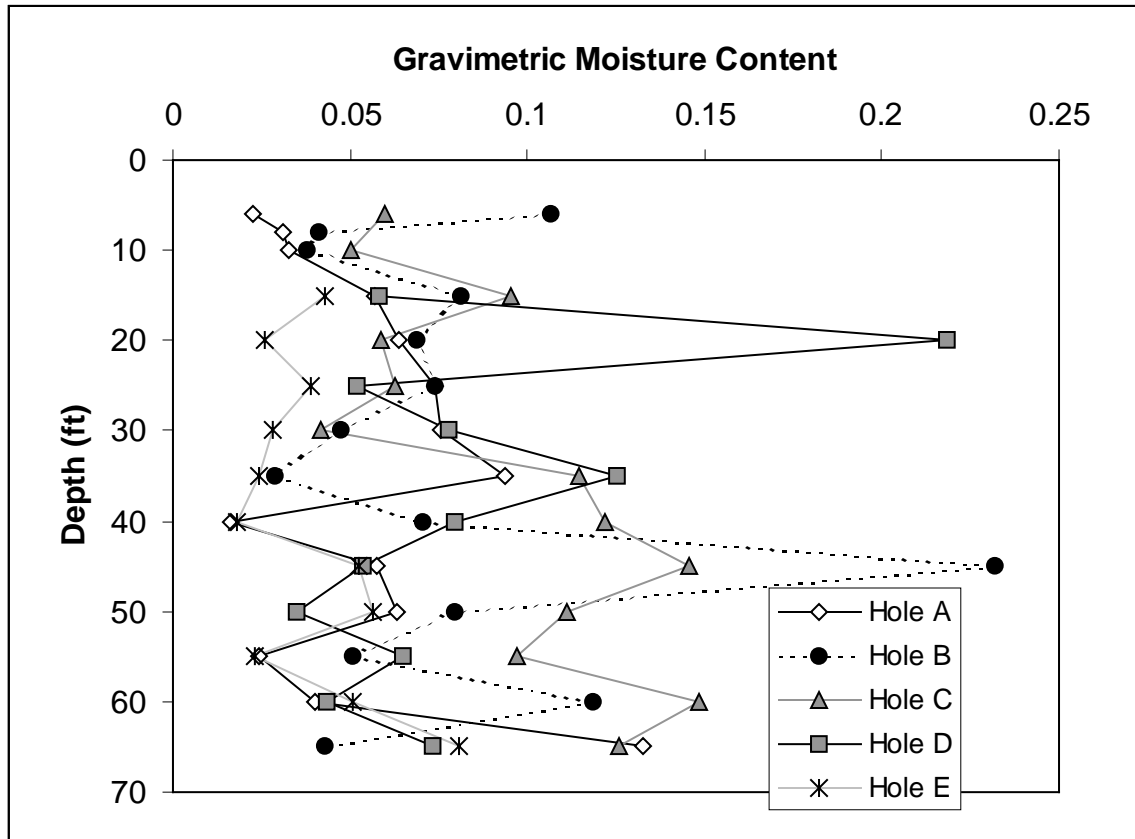
Figure 0-8. Gravimetric moisture content as a function of depth for all bore holes.
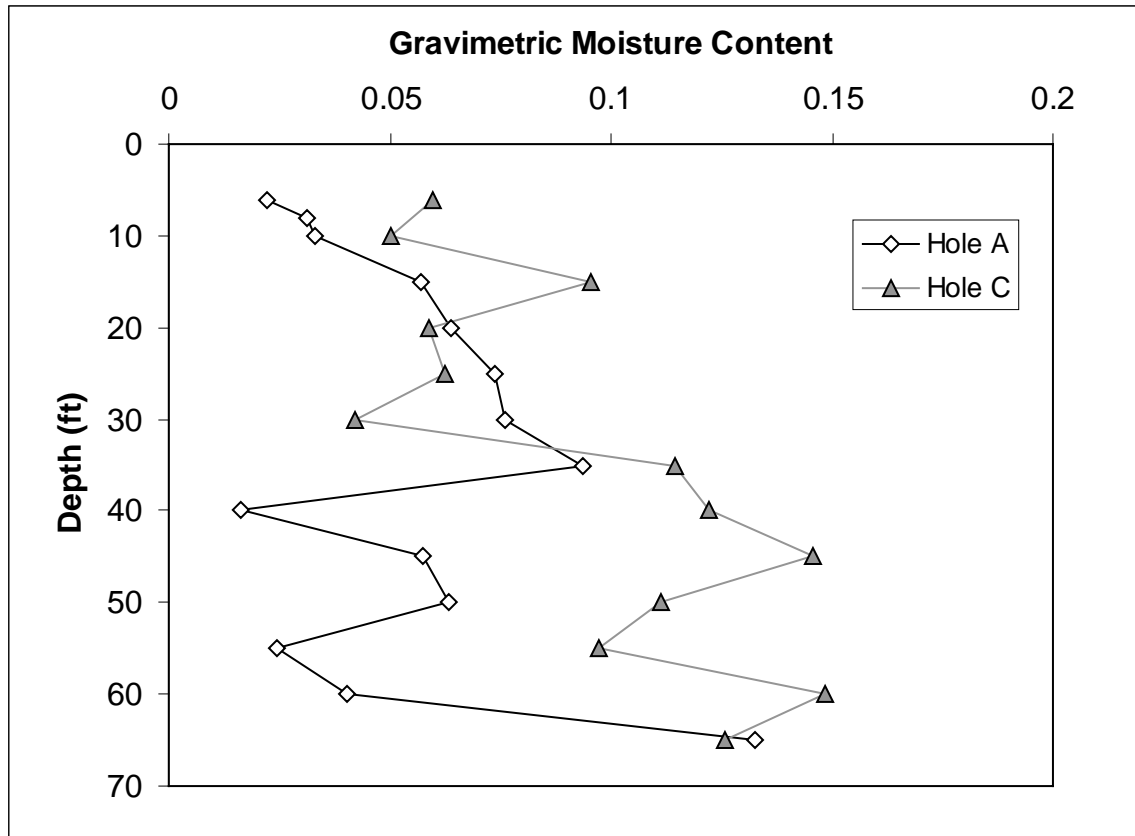
Figure 0-9. Gravimetric moisture content as a function of depth for bore holes A (background) and C (closest to absorption field).

1.13. Results of Vadose Zone Geochemical Investigations

The soil samples from the bore hole sampling program were subjected to a deionized water leach process as described in Section 3.2 to recover soluble constituents from the soil and associated pore water. The leachate was subjected to chemical and microbiological analysis. The analytical results could then be combined with the measured water content of each soil sample to allow determination of the concentration of each constituent in the pore water of the unsaturated soils. The pore water solute concentration is calculated from the concentration of each constituent in the leachate and the water content of the soils by using the following equation:

$$C_{\text{pore water}} = \frac{V_{\text{leachate}} C_{\text{leachate}}}{M_{\text{soil}} W_{\text{soil}} \rho_{\text{H2O}}}$$

(4-1)

where

V_{leachate} = the volume of deionized water leachate

C_{leachate} = the constituent concentration in the leachate,

M_{soil} = mass of the soil sample

W_{soil} = the water content of the soil sample

ρ_{H2O} is the density of water (1 g/mL)

Note that the solute concentration in the pore water depends on the inverse of the soil water content (W_{soil}). Note further that due to the coarseness of the soils at this site W_{soil} is small and highly variable (Figure 0-8). This introduces significant variability to the following results.

The purpose of this testing was to assist in developing an understanding of the transformations and transport of the constituents of the wastewater infiltrating into the subsurface soils from the absorption field. This section describes the results of these analyses. To facilitate analysis and interpretation of the vadose zone sampling results, only the results for the background hole (hole A) and the hole closest to the beginning of the absorption field (hole C) are presented graphically. Complete analyses and plots of data from all bore holes are contained in Appendix 7.

1.13.1. Total Nitrogen, Ammonia, and Nitrate

The nitrate concentration in pore water from the soil samples is presented graphically as a function of depth in Figure 0-10. These results indicate that Hole C, which is closest to the beginning of the leach field, has consistently elevated nitrate values compared to the values from the other holes, especially in the samples from the bottom 20 ft of the hole. The elevated nitrate concentrations at shallow depths, particularly in hole A, are believed to be due to fertilizer that was applied to the surface to promote vegetative growth.
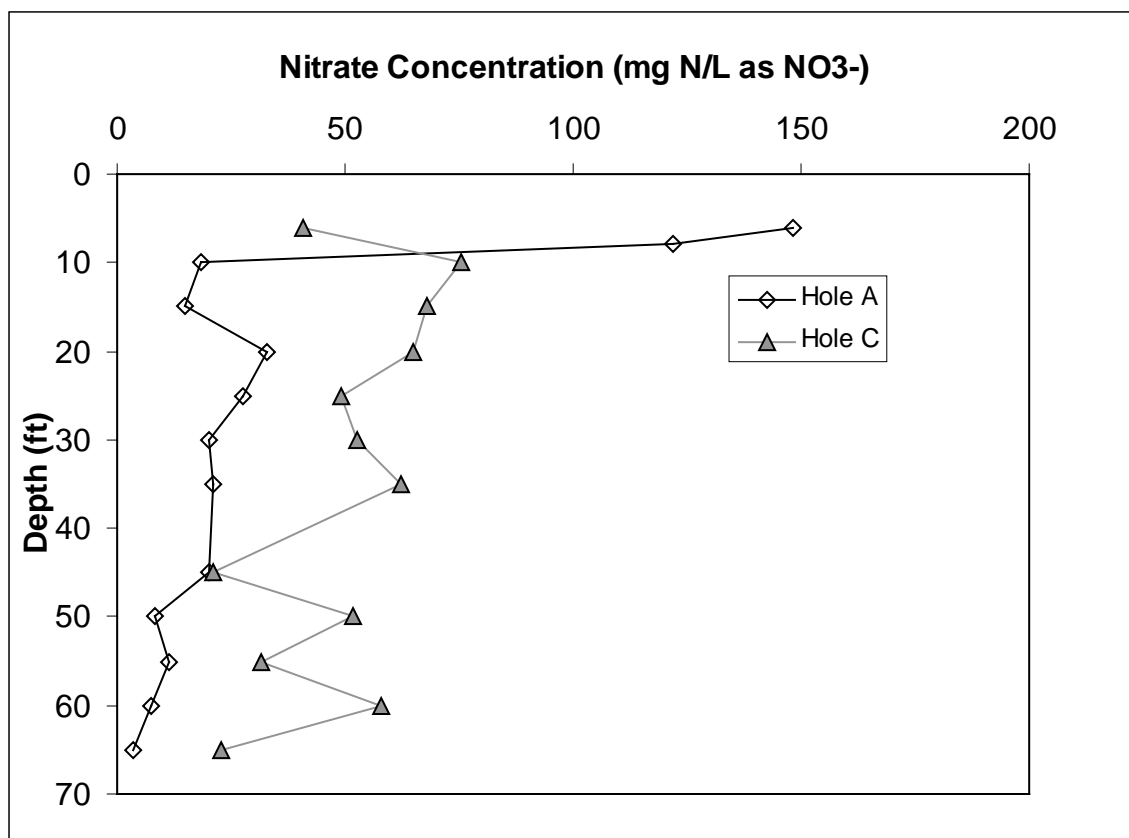
Figure 0-10. Pore water nitrate concentrations in unsaturated soil samples (mg N/L as NO3-).

Ammonia concentrations in pore water from the soil samples are given in Figure 0-11. Although ammonia was detected in these samples, most was at very shallow depths in the immediate vicinity of the drain pipes in (Holes C and E). This is representative of untreated septic tank effluent. The presence of nitrate and absence of ammonia in deep soil samples indicates that nearly complete and rapid nitrification occurs in the effluent as it migrates vertically through the soil column. This is consistent with the conceptual model of nitrogen transformations and transport presented in Figure 0-1 in which cationic ammonium (NH_4^+) is sorbed onto soil particles near the absorption field and microbially oxidized to nitrate in the aerobic conditions of the unsaturated soil.
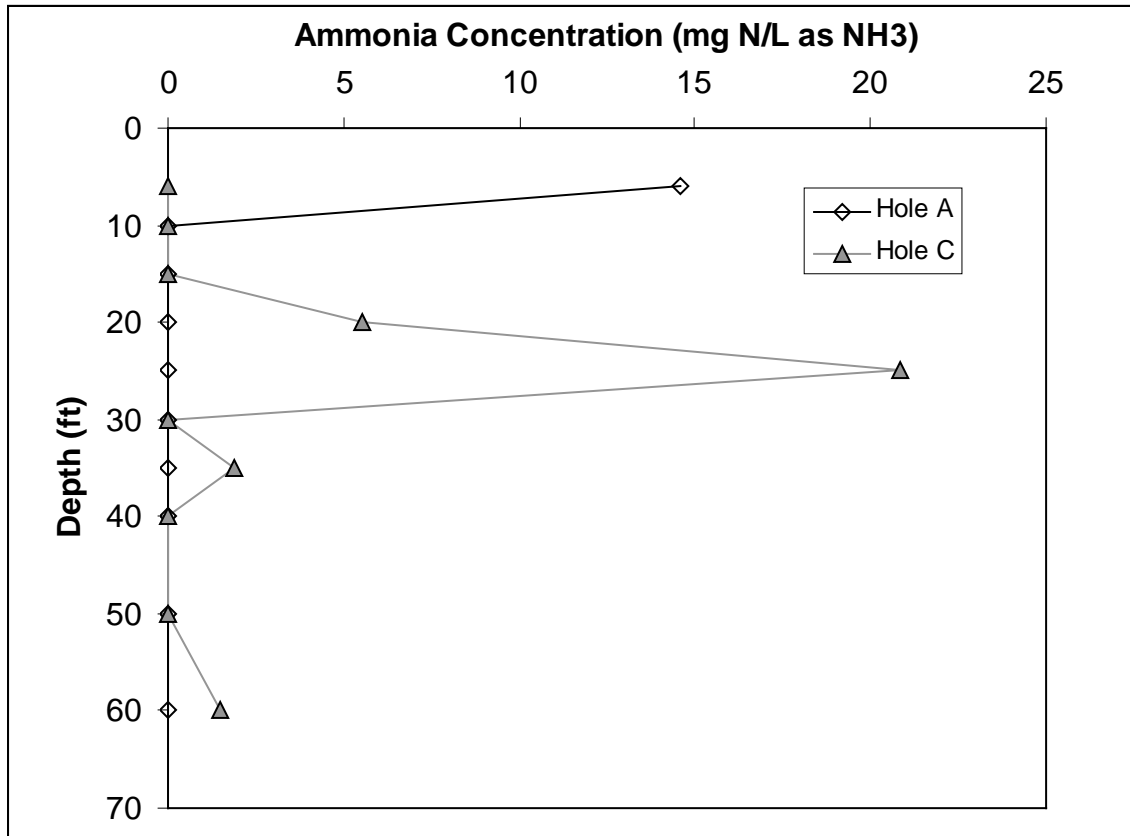
Figure 0-11. Pore water ammonia concentrations in unsaturated soil samples (mg N/L as NH3).

The total nitrogen concentration (mg N/L) in the soil sample pore water is presented in Figure 0-12. The total nitrogen content represents the sum of the concentrations of the organic nitrogen species, ammonia, nitrite, and nitrate. Generally, the total nitrogen concentration is greater than the nitrate concentration which is consistent with expectations and provides some confirmation of the validity of the analytical procedures. This result is significant because the nitrogen species were measured by two different techniques.
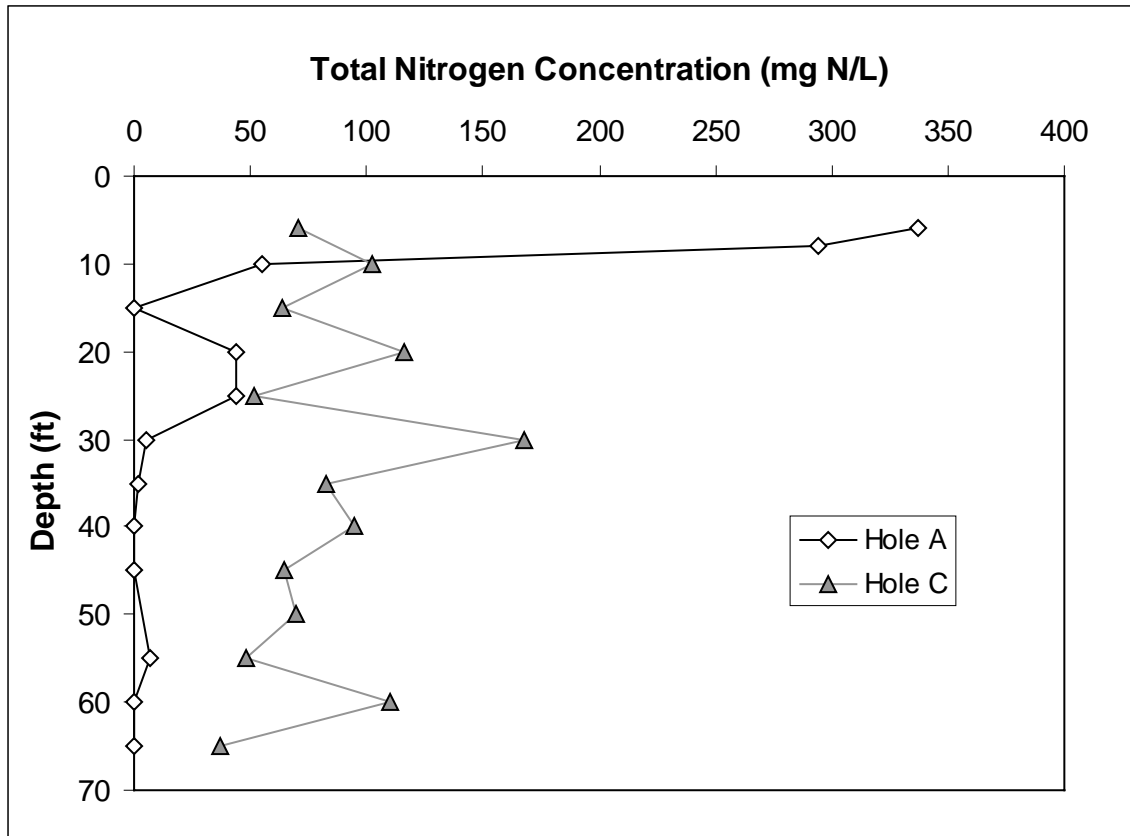
Figure 0-12. Pore water total nitrogen concentrations in unsaturated soil samples (mg N/L).

1.13.2. Other Anions

Phosphate was not present in any of the samples except at trace concentrations. Phosphate forms insoluble precipitates with a number of common soil cations including calcium and iron and generally does not migrate through saturated or unsaturated soils. A pulse of potassium bromide (KBr) was introduced to the wastewater system two weeks prior to the soil boring program. Bromide was detected at low concentrations only in those holes near the beginning of the absorption field (holes C and D). The purpose of its addition was to serve as a conservative tracer that could be used to measure the vertical contaminant migration velocity of water through the unsaturated zone. The calculated values of bromide in the soil pore water for bore hole C, based on the moisture content of each sample, is displayed in Figure 0-13. The apparent spike in the bromide concentration at 10 ft depth may be due to the low measured water content of this soil sample (see Figure 0-9) rather than an actual increase in the bromide level as

the bromide concentrations in the leachate from all of the soil samples were nearly uniform at concentrations ranging from 1 to 2 mg/L. This value is just above the estimated detection limit of 0.5 mg/L. Similarly, the apparent decrease in bromide concentration below 35 ft depth may be due to the measured increase in soil water content at this depth. The fact that bromide was detected at nearly constant concentration throughout the 65 ft soil column and only in the bore holes nearest the absorption field indicates that the pore water moves rapidly through the soil at a rate of at least 65 ft/14 days, or 4.6 ft/d
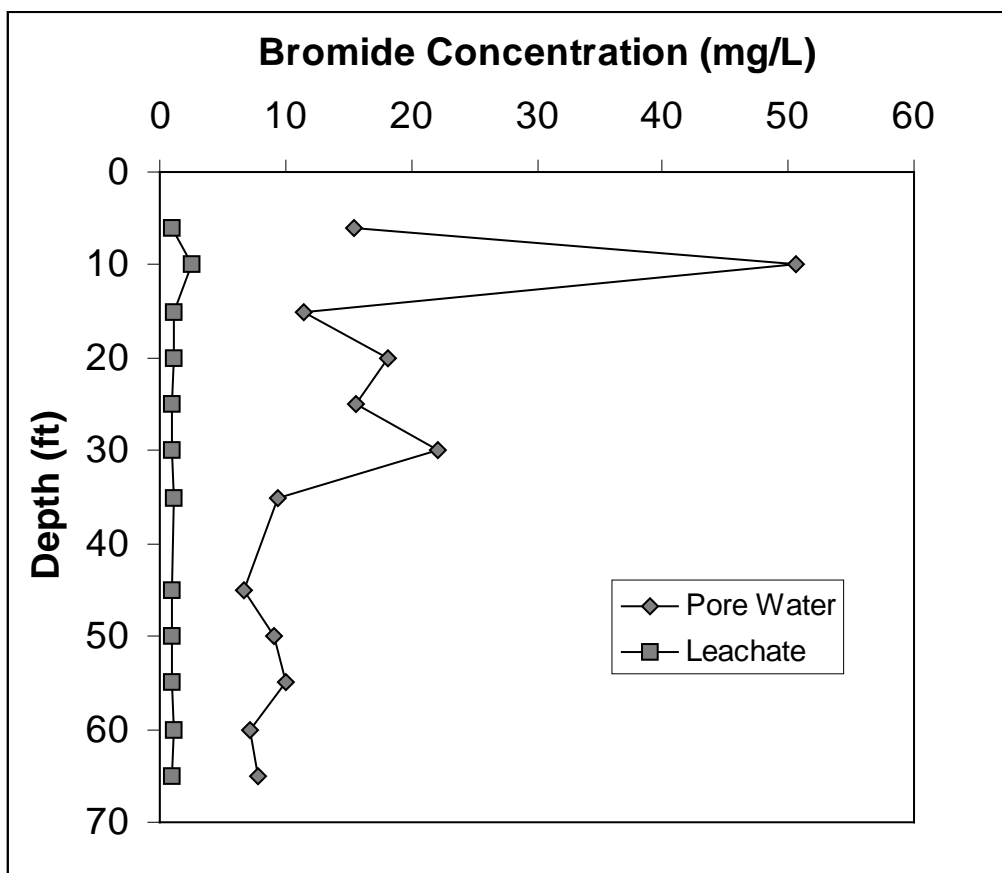
Figure 0-13. Bromide concentration vs. depth in hole C.

Chloride and sulfate concentrations in the pore water are given in Figure 0-14 and Figure 0-15. The greatest values tend to be in hole A, with hole C having the lowest values. The low values in hole C are believed to be due to the dilution effect associated with continuous addition of wastewater to the absorption field. Higher chloride concentrations in the background holes

(hole A) are believed to be due to the concentration effects of evapotransporation which removes water from the soil but leaves the ions behind. A similar effect is noted for the sulfate concentrations.
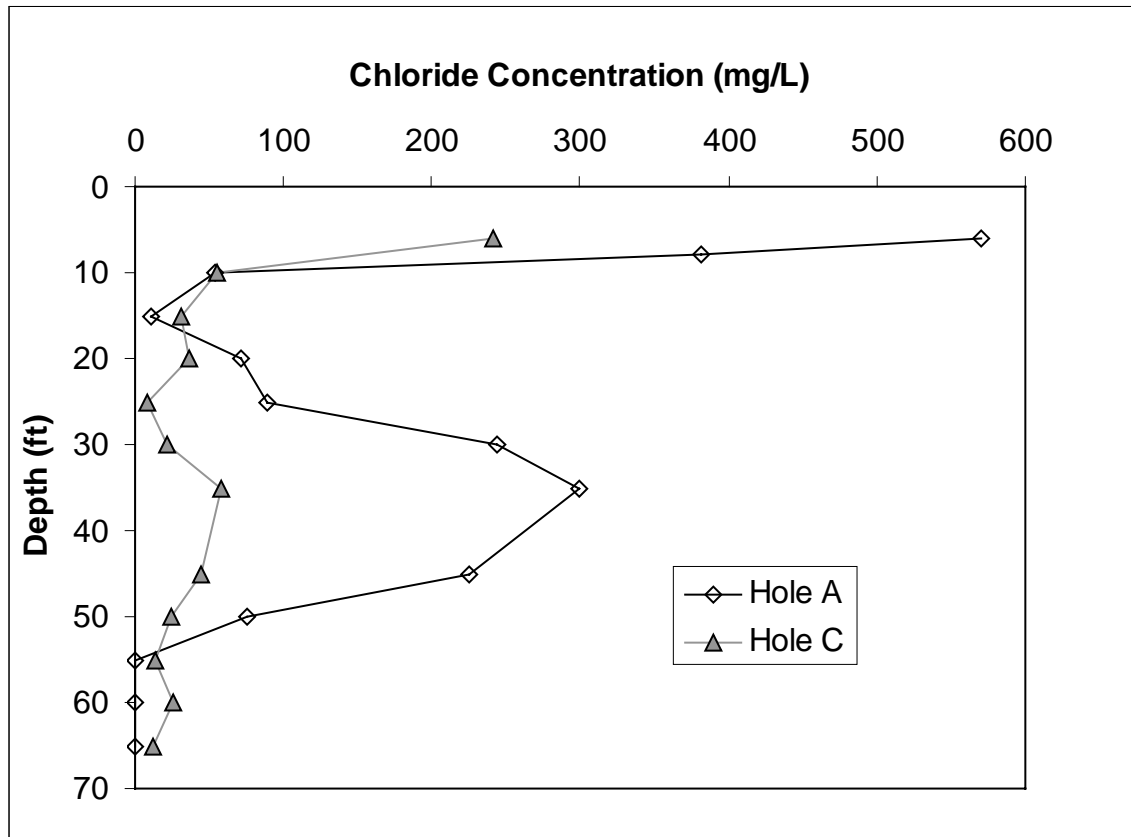
Figure 0-14. Pore water chloride concentrations in unsaturated soil samples (mg/L).
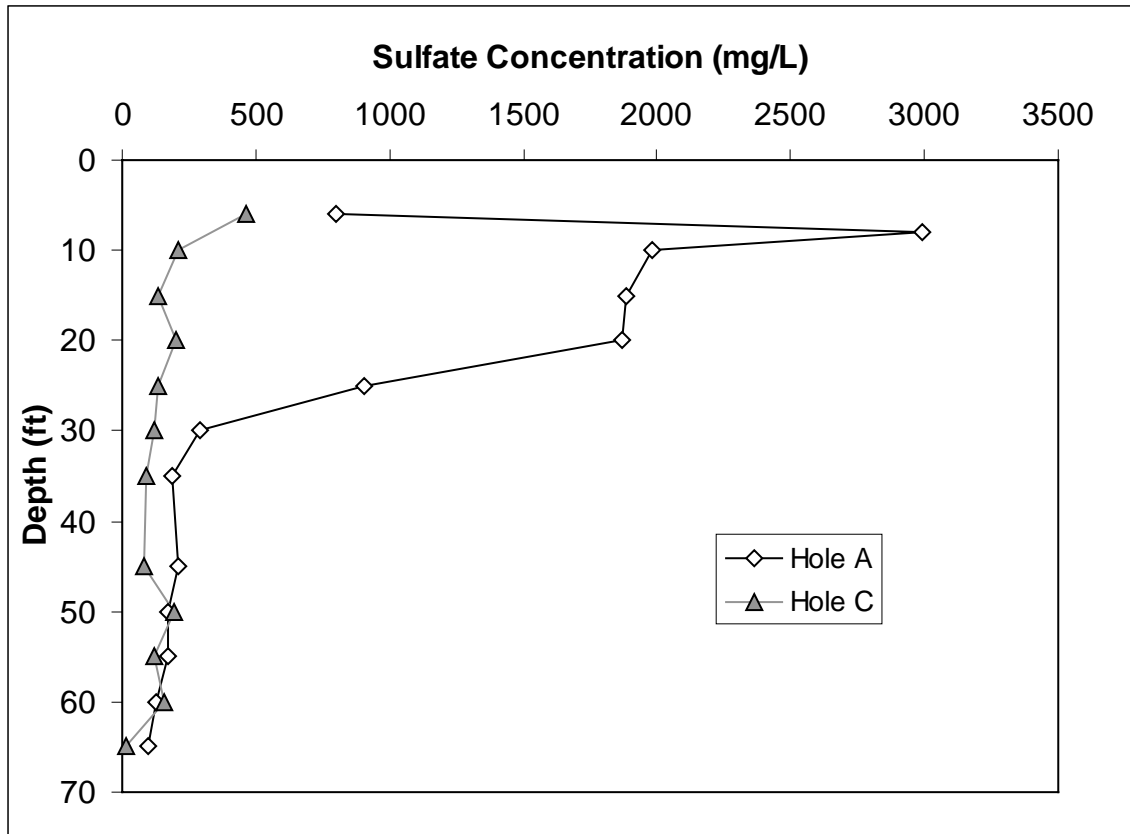
Figure 0-15. Pore water sulfate concentrations in unsaturated soil samples (mg/L).

Figure 0-16 is a plot of the pore water chloride versus sulfate concentrations. This plot graphically illustrates the different pore water chemistry of soils from the background hole (hole A) and the bore hole nearest the beginning of the absorption field (hole C). The pore water in the background hole has widely varying concentrations of both anions, and their ratio is not constant as seen by the scatter on the chart. In contrast, the chloride and sulfate concentrations in hole C are very tightly grouped, the one exception being the sample from the very top of the soil column which is likely strongly influenced by evapotranspiration effects. The data suggests that the continuous application of wastewater to the leach field has washed the soil of the high chloride and sulfate concentrations that were originally present. Figure 0-16 provides confirmation that hole A is indeed a background hole that has not been influenced by discharge from the absorption field.
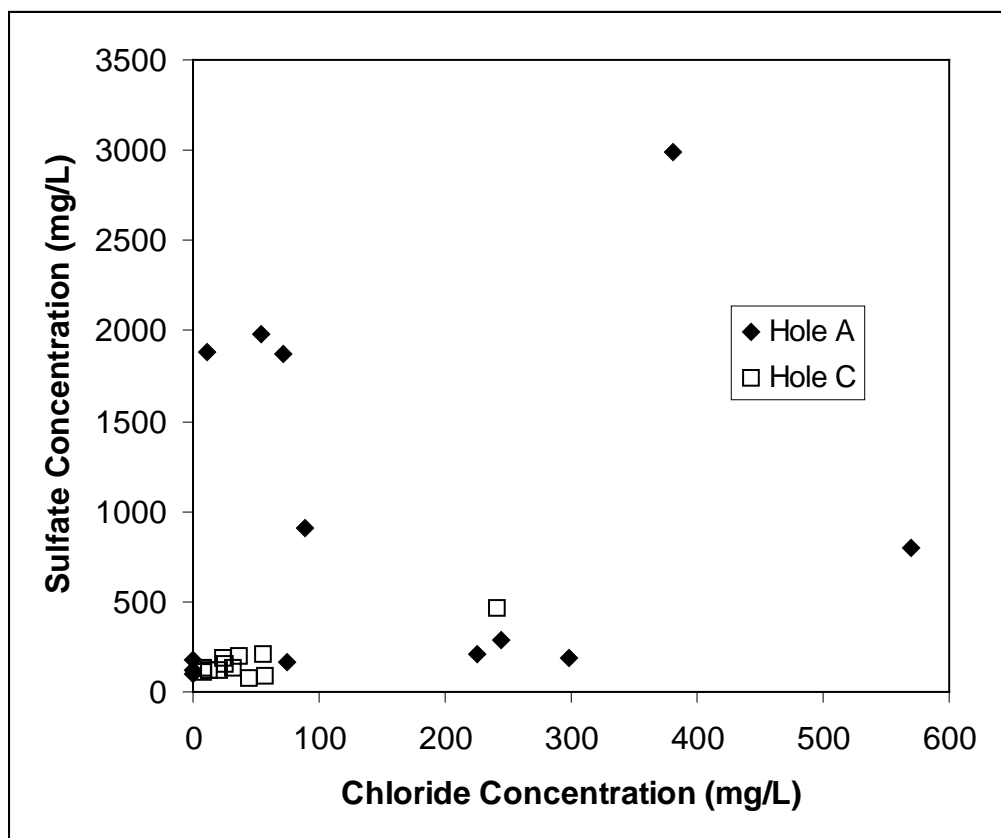
Figure 0-16. Comparison of chloride versus sulfate concentrations for bore holes A and C.

Figure 0-17 is a plot of the concentration of the pore water chemical oxygen demand

(COD) versus depth. As shown by the other plots, there is a lot of variability in this data which

is due in large part to the variation in soil water content. There are two remarkable phenomenon

revealed by this figure. First, the pore water COD concentrations in all of the holes are quite

high. Whereas the COD of septic tank effluent is reported to be in the range of 680-730 mg/L

(USEPA, 1980), the pore water concentrations calculated from the COD values measured in the

soil leachate are 5 to 10 times this value. The second feature of interest is that the COD values in

the background hole A is substantially greater than in hole C, which provided soil samples

nearest the beginning of the leach field. There are two factors which contribute to these results.

First, the high COD concentrations in the background soil samples are believed to be due to

concentration of organic compounds by evapotranspiration over very long time periods, possibly

extending over centuries or longer. In contrast, the soil beneath the drainfield is continuously

replenished by leachate from the septic tank. Therefore, the pore water COD values are similar

to those found in poorly treated wastewater and are not significantly concentrated by evaporation of water. Second, the low water content and very low total nitrogen content of the unsaturated soil will almost certainly limit growth of heterotrophic organisms. Thus, while there appears to be a supply of organic carbon in the background soils, microbial growth is limited by other factors, principally the low concentration of nitrogenous nutrients and the low soil moisture content. Finally, the soils in hole C have all of the conditions necessary for rapid growth of heterotrophic bacteria including adequate soil moisture, a continuous source of nutrients, and high concentrations of degradable organic carbon substrate. The data in Figure 0-17 shows a trend of decreasing COD concentration in the soil pore water which suggests that this organic substrate is being consumed as it passes through the soil column.
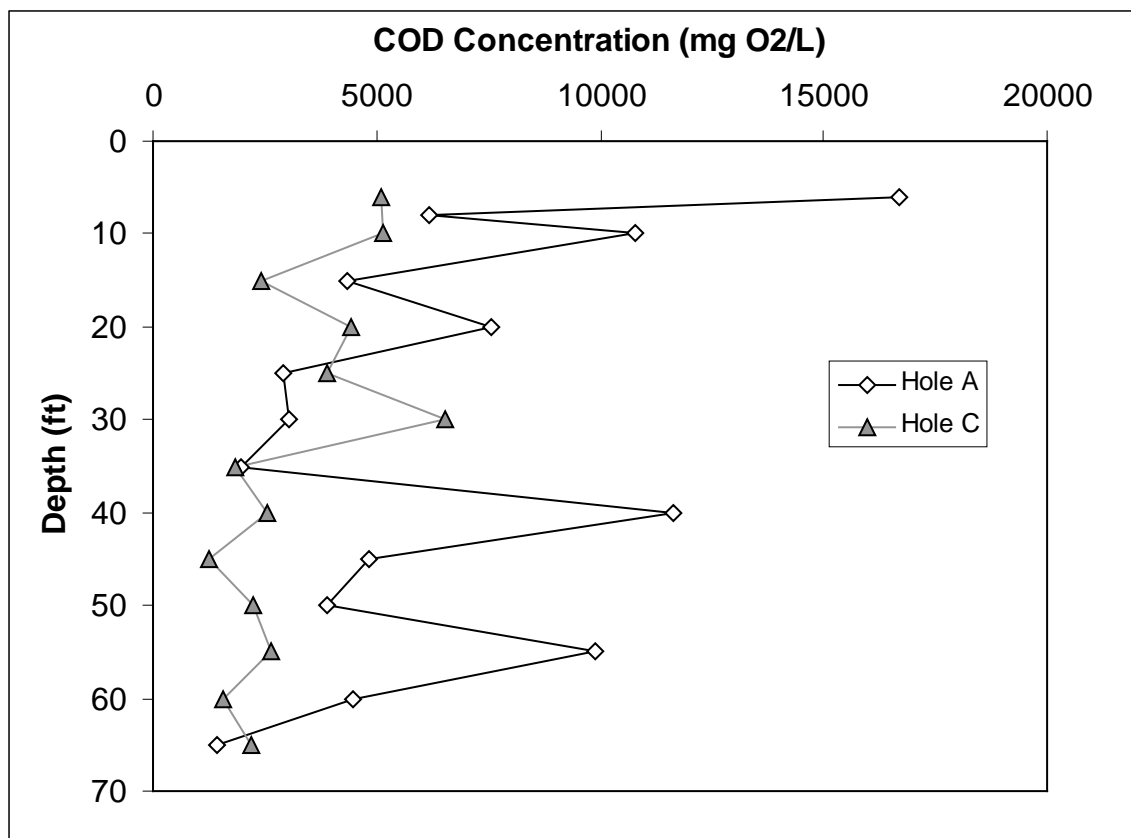
Figure 0-17. Pore water chemical oxygen demand (COD) concentrations in unsaturated soil samples (mg/L).

The high COD values in the background hole (hole A) suggest a possible dynamic situation beneath a new absorption field in a thick vadose zone that might result in denitrification. A population of nitrifying organisms would quickly become established in the well aerated soil immediately beneath the absorption field following start up of the disposal system. However, if the native soil has a sufficiently high concentration of organic material, these nitrates would be reduced to $N_{2(g)}$ in deeper formations by denitrifying organisms utilizing the naturally occurring organic carbon in the deep soil as their energy source. This would have the effect of delaying the appearance of the nitrate plume in the water table aquifer until all of the residual organic carbon had been consumed. Based on the results obtained in this study it is not possible to evaluate the probability of such a phenomenon, but conditions seem suitable for its occurrence.

1.13.3. Microbial Results

Microbial analyses were performed on deionized water leachate from selected soil samples from hole C as described in Section 3.2. These results are given in Table 0-7 for fecal coliform bacteria and total heterotrophic organisms. There was a complete absence of fecal coliform bacteria in all of the soil samples. This finding is consistent with that of Amjad et al. (1993) who found no coliform organisms within a few feet of septic seepage pits in Mojave desert soils. This suggests that it is unlikely that pathogenic microorganisms will reach the water table in the well sorted soils at this site given the long distances between the leach field and the water table.

	Conc. (No. orgs./mL)	
Depth (ft)	Fecal coliform	Total Heterotrophic bacteria
6	0.	0.0
10	0.	1,100.
30	0.	300.
65	0.	0.0

Table 0-7. Bacterial counts versus depth in hole C.

1.13.4. Discussion of Vadose Zone Sampling Results

The results of the vadose zone sampling program show that water flow through unsaturated soils in the North Albuquerque Acres area is rapid due to their high sand content and associated high porosity. Furthermore, all of the chemistry supports the hypothesis that the soils are well aerated and chemical transformations are those expected to be mediated by aerobic microbial populations, oxidation of organic carbon and reduced nitrogen species. The presence of oxidizing conditions results are consistent with the measured values for soil moisture content. In other words, it is evident that septic tank effluent has moved downward noticeably beneath hole C beyond the deepest sample retrieved (65 ft). Given that the Sheriff's substation had only been in operation for 14 months at the time of sampling, this suggests that the downward velocity of the effluent is at least 60 ft/yr (i.e. the wetting front has passed through at least 65 ft in 14 months). This is considered to be the lower bound of the estimated vertical infiltration rate. The bromide tracer results indicate that a conservative tracer (the bromide) migrated to a depth of at least 65 ft in 14 d. This corresponds to a vertical water flow rate of approximately 1700 ft/yr. In the absence of significant lateral spreading due to matric potential gradients, this velocity suggests the time for septic tank effluents to reach the water tables ranges from less than 1 year (vertical flow rate of 1700 ft/yr) to about 10 years (vertical flow rate of 60 ft/yr).

These results also reveal that elevated nitrate concentrations are present in deep vadose zone soils, and that this chemical species migrates readily with the downward movement of water. There is no indication from these data that denitrification is occurring at depth, however, the presence of very high COD values in pore water samples from the background hole suggest that this may be possible at depths greater than those sampled in this investigation.

In considering the results of the vadose zone sampling program it is important to recognize that due to limitations associated with a hollow stem auger drill rig, soil samples were only collected from the top 65 feet of the soil column. Depth to ground water at this site is estimated to be approximately 500 feet, therefore, definitive conclusions regarding possible transformation and transport of contaminants at greater depths are not possible.

MODELING STUDIES

1.14.   Ground Water Flow and Contaminant Transport Model

To assist in predicting nitrate concentrations in ground water underlying an unsewered subdivision a numerical model was developed using the two dimensional U.S. Geological Survey Method-of-Characteristics (MOC) Model for Solute-Transport (Konikow and Bredehoeft, 1984) for the Sandia Heights area. MOC uses the finite difference ground water flow model MODFLOW (McDonald and Harbaugh, 1988) to model the ground water flow regime, then the method of characteristics is used to calculate solute transport. The modeling results predict the movement of contaminants throughout the aquifer as a function of time.

Figure 0-1 is a graphical overlay of the modeled area onto an aerial photo of the Sandia Heights area. The inner rectangle in Figure 0-1 is the region where the on-site disposal systems are concentrated as a result of development in Sandia Heights area. A conceptual diagram of the Sandia Heights model is shown in Figure 0-2. The aquifer was modeled as a single layer, unconfined formation in the alluvial deposits of the upper Santa Fe formation located at the base of the western face of the Sandia Mountains. Three formation thicknesses were used to simulate the aquifer in the regions separated by the two mountain front faults that dissect the area from north to south. These thicknesses were estimated from the well record for the up gradient well, the well log for SH02, and data developed by Hawley and Haase (1992). The easternmost segment of the model is assumed to be perched on top of granite bedrock and to have a thickness of about 40 feet. In actuality, this is a wedge of sediments ranging from zero to various values in thickness on top of the down dropped eroded granite surface. The thickness of the next two segments was set to 300 and 500 feet to the increased thickness of unconsolidated deposits on the western side of the two faults. The actual thickness of this part of the aquifer is much greater, especially in the western most segment, where it is thousands of feet thick (Hawley and Haase, 1992). However, horizontal layering of the deposits noted by Hawley and Haase (1992)

Waste-water recharge area

SH02

SH01

Paseo del Norte Blvd

EG01

SH03

mway Blvd
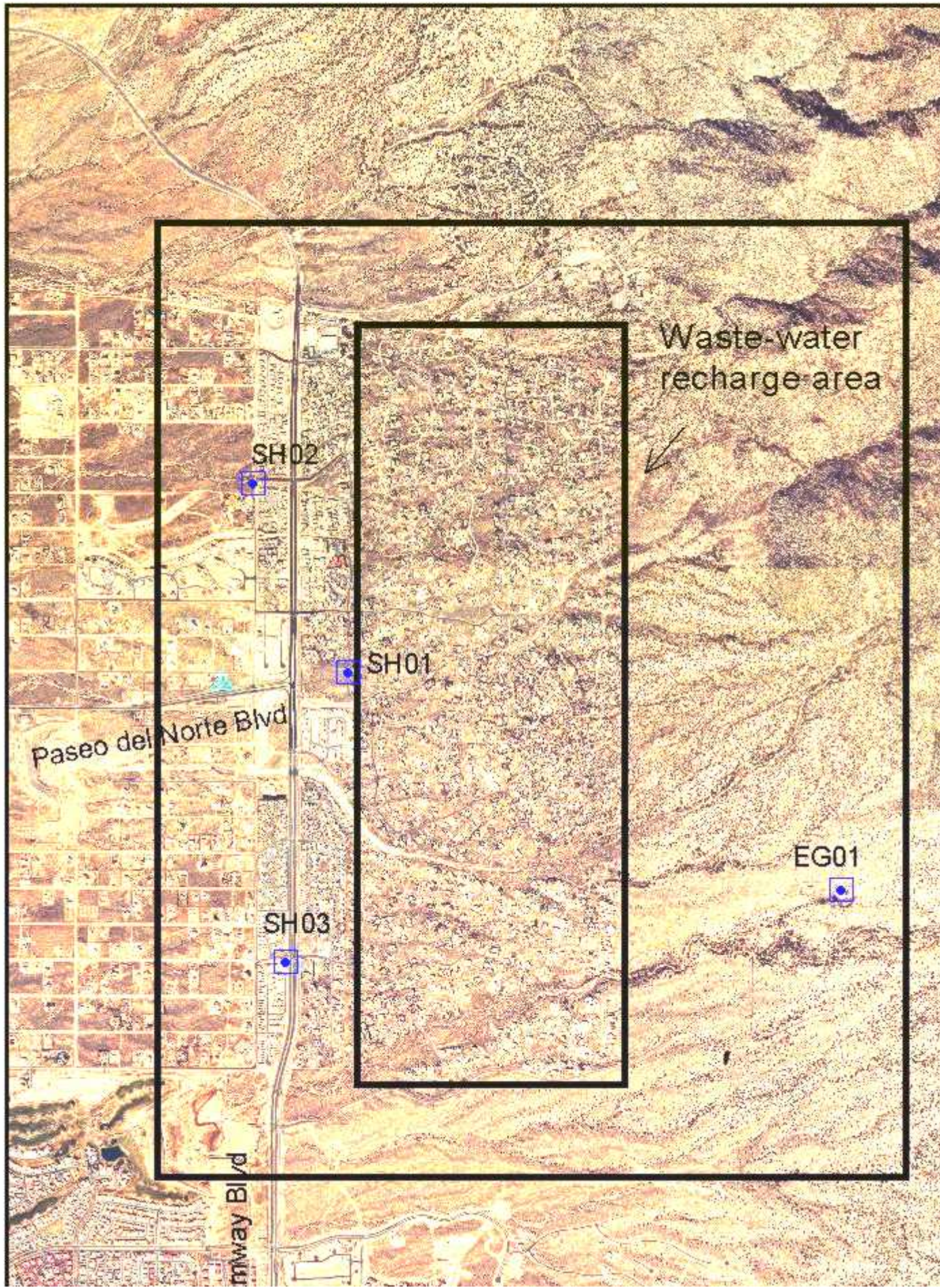
Figure 0-1. Aerial photo of model area (1996) with model boundaries and monitoring wells shown.

greatly reduces the vertical hydraulic conductivity in comparison to the horizontal hydraulic conductivity, therefore, the thickness of the formation used in this portion of the model is of little importance. The effect of on-site treatment systems was simulated by including a recharge factor in each grid cell that represents the flow and water quality characteristics associated with all of the absorption field systems in that cell.
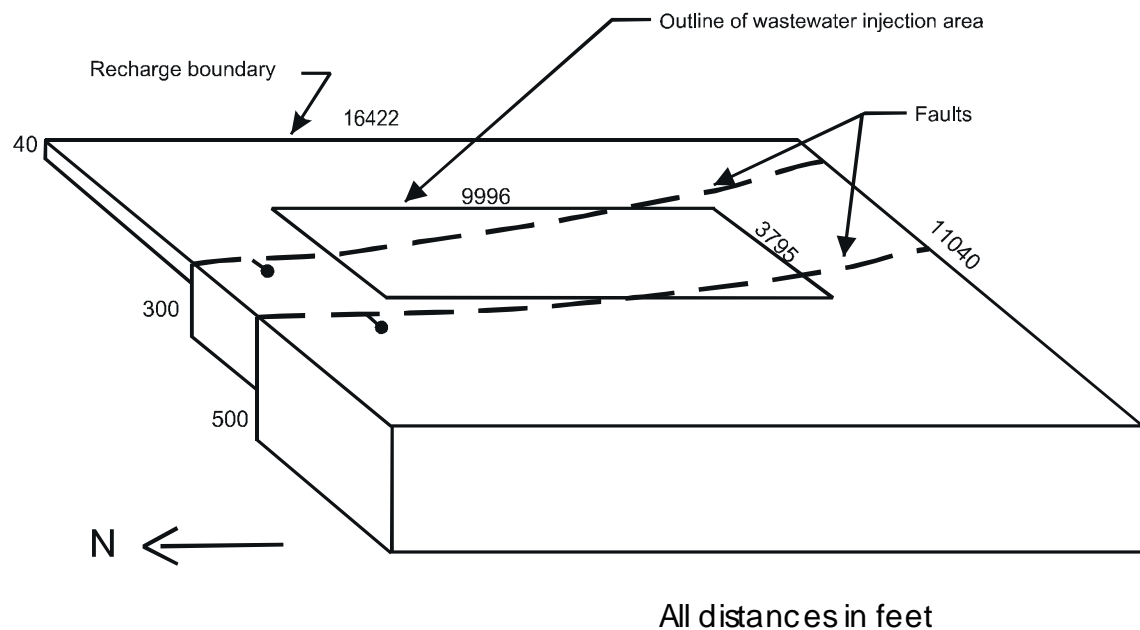
Figure 0-2. Diagram of the conceptual model of ground water flow model for the Sandia Heights area.

1.14.1. Boundaries, Recharge, and Calibration

Two types of boundaries were used in the ground water flow model, specified flux and constant head. The eastern and western boundaries of the model are specified flux boundaries. The flux at the eastern boundary simulates the mountain front recharge by placing injection wells in each of the eastern boundary cells. Pumping wells in each western boundary cell simulate the western boundary flux. For a steady state model to run the head must be defined in at least one cell in order to provide an initial starting point for the iterations. Therefore, one constant head value was assigned to the cell corresponding to the up gradient monitoring well location. The north and south boundaries are no flow boundaries. The model has no precipitation recharge.

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The mountain front recharge per cell was calculated from a recharge value of 12.21 ft³/s as contained in the USGS model of the Albuquerque basin (Kernodle, et. al., 1994, Figure 5). The boundary length of the part of the USGS model is along the western face of the Sandia Mountains is approximately 52,800 feet and each model cell boundary is 714 feet long. Therefore, the mountain front recharge per cell is equal to:

$$\frac{714 \text{ ft}}{52,800 \text{ ft}} \times 12.21 \text{ft}^3/\text{s} = 0.165 \text{ ft}^3/\text{s per cell} \qquad (5.1)$$

The value 0.165 ft³/s was used as a well injection rate for each boundary cell along the eastern boundary of the model.

The ground water flux along the western boundary of the grid was used to calibrate the model by adjusting the pumping rate in each cell until the model output heads corresponding to the location of the monitoring wells matched the heads measured in the monitoring wells installed for this project.

Discharge from the onsite wastewater disposal systems was modeled by adding an injection well to each cell in the Sandia Heights area. The injection rate was calculated as:

[Avg. Discharge Per System] x [No. of Lots per Grid Cell] = Well Injection Rate   (5.2)

Using .422 acre-ft/yr as a discharge value per septic tank from the annual return flow credit granted to Sandia Peak Utilities by the Office of the State Engineer, the injection rate for each cell is calculated as follows:

$$[.422 \text{ acre - ft/yr}] \times \frac{1}{365 \text{ d/yr}} \times \frac{1}{86,400 \text{ s/d}} \times 43,560 \text{ ft}^3/\text{acre - ft} \times 7 \text{ lots/cell} = .0041 \text{ ft}^3/\text{s} \qquad (5.3)$$

A value of 34.1 g N/d as total nitrogen was used for the septic tank effluent based on a previous report by CH2M-HILL (1990). This yields a nitrogen concentration in the injection wells of:

$$\frac{34.1\,\text{g/d x}\,\dfrac{1}{86{,}400\,\text{s/d}}\,\text{x 7 septic tanks/cell}}{0.0041\,\text{ft}^3/\text{s}} = 0.674\,\text{g/ft}^3 \qquad (5.4)$$

This value, $0.674\,\text{g/ft}^3$, is equal to a nitrogen concentration of 23.8 mg N/L. In this modeling exercise this nitrogen is assumed to be present as NO_3^-.

There were two important assumptions incorporated in the contaminant transport modeling. First, The MOC model (Konikow and Bredehoeft, 1984) used in this investigation does not include algorithms allowing simulation of contaminant transformations such as the oxidation of NH_4^+ to NO_3^-. Furthermore, there is no data to permit estimation of the rate of this or other biologically mediated nitrogen transformations (e.g. denitrification). Therefore, all of the nitrogen discharged to ground water was assumed to be in the form of nitrate and it was assumed to be a conservative tracer compound that migrates at the same velocity as the ground water and does not participate in any biological or chemical reactions which would result in natural attenuation. This assumption is consistent with the conceptual model of nitrogen transport through the vadose zone presented in Chapter 2.

The second assumption incorporated in the contaminant transport modeling is that the nitrates are continuously and directly applied to the ground water through an imaginary injection well added to each cell of the model. In reality, nitrogen species from the on site wastewater disposal systems must first pass through the unsaturated soil column, be oxidized by nitrifying bacteria in the soil, and then migrate vertically to the water table. As discussed in Chapter 4, this is estimated to take between 1 and 10 years and depends on the depth to the water table and the hydraulic characteristics of the soil. Another limitation of this modeling approach is that the Sandia Heights subdivision has developed over a period of approximately 20 years. Therefore, instead of a step increase in the contaminant application rate, housing development has increased continuously in the area over this period of time to its present density of approximately one house per 0.8 acres. The assumption of a step increase in the nitrogen loading rather than a continuous increase will affect the predicted shape of the contaminant arrival distribution curve at the down gradient monitoring wells.

The hydraulic properties of the aquifer used in the model were the same as reported by McAda, et al, (1994). The hydraulic conductivity was obtained from a slug test on a monitoring

well for a subdivision approximately one mile south of Sandia Heights performed by Geohydrology, Inc., and additional data were derived from the geohydrologic description of the Albuquerque Basin by Hawley and Haase (1992). The number of onsite wastewater disposal systems in the model area was taken from the annual return flow credit report by Sandia Utilities to the New Mexico Office of the State Engineer. This report includes the number of houses with septic tank systems, the total ground water withdrawal by the water supply utility, and the estimated recharge to the aquifer by infiltration from the septic tanks. The City of Albuquerque provided Geographical Information System (GIS) data on the lot size and number of unsewered connections. Table 0-1 lists all of the input parameters used for the Sandia Heights ground water flow model and their sources.

Parameter	Value	Input Value	Reference
Hydraulic Conductivity	4 ft/d	4.63e-5 ft/s	USGS (McAda) MRGB Workshop 11 Feb 98
Hydraulic Gradient	.11	0.10	
Saturated Thickness	40 - 500 ft	40 - 500 ft	
Recharge (Natural)	12.21 ft^3/s	.165 ft^3/s	USGS Report 93-4149 fig 39 p. 86
Recharge (Return Flow per Septic tank)	.422 acre-ft/yr (Sandia Utilities value)	-.0041 ft^3/s	Sandia Properties, Ltd, 1996 OSE Usage Report
Aquifer Dispersivity	100	100	CH2M Hill, '92
Aquifer Porosity	.2	.2	
Nitrogen Loading	34.1 g/d as N (per Septic System)	0.674 g/ft^3 per cell (23.8 mg/L)	CH2M Hill. '92
Aquifer Homogeneity	1/1 Horizontal	1/1	CH2M Hill, '92
Background Nitrogen Concentration	< 1 mg/L	0	Elena Gallegos well analyses results
Modeled Area	103458600 ft^2 (3.7 mi^2)		
Cell Area		246,330 ft^2	
Cell Dimensions (x,y)		714 ft -345 ft	
Number of Septic Systems	1094		Sandia Properties, Ltd, 1996 OSE Usage Report
Mean Lot Size	35,548 ft^2		COA GIS
Number of Septic Systems/Cell		7	
Array Dimensions		23 x 32	
Pumping Wells	2 Sandia Properties Ltd. Co. supply wells	1 pumping well in cell 12, 25	OSE return flow summaries for 1996
Pumping Rate (ft^3/s) combined avg annual rate	1.27	1.27	OSE Return flow summaries for 1996
Time Period		30 years	

Table 0-1. Input parameters used for the solute transport flow modeling.

1.14.2. Modeling the Faults

Incorporating the effects of the two faults that extend from north to south across the Sandia Heights subdivision is important to the accuracy of the ground water flow model. It is believed that these faults act as barriers to westerly ground water flow, thus, in order to simulate the water table drop across the two faults, they were modeled as zone of low hydraulic conductivity. No data is available regarding the hydraulic conductivity of formation in the immediate vicinity of these faults. Accordingly, an analytical approach was used to estimate the K values.

The conceptual model of ground water flow in the Sandia Heights area is shown in a three dimensional perspective view in Figure 0-2 and a cross section view is presented in Figure 0-3. This illustrates that the depth to bedrock increases with increasing distance westward from the Sandia Mountains as has been shown by Hawley and Haase (1992). Each fault acts as a low conductivity barrier to ground water flow. The total flow (Q) in each zone is the same, but since the saturated thickness increases to the west, the hydraulic gradient decreases.

Because the MOC code is a two-dimensional model, selection of the thickness of each region in the model (Figure 0-3) will effect the results in that the contaminant will be distributed uniformly throughout the vertical profile of the aquifer. The extent of vertical mixing of contaminants in the Upper Santa Fe formation is not known.

West

East

Fault Zone

Fault Zone

x5

x4 ▶ x3

Land Surface

x2 ▶ x1

x0

Monitoring Well

Mountain Front Recharge

Water Table

b = 40 ft
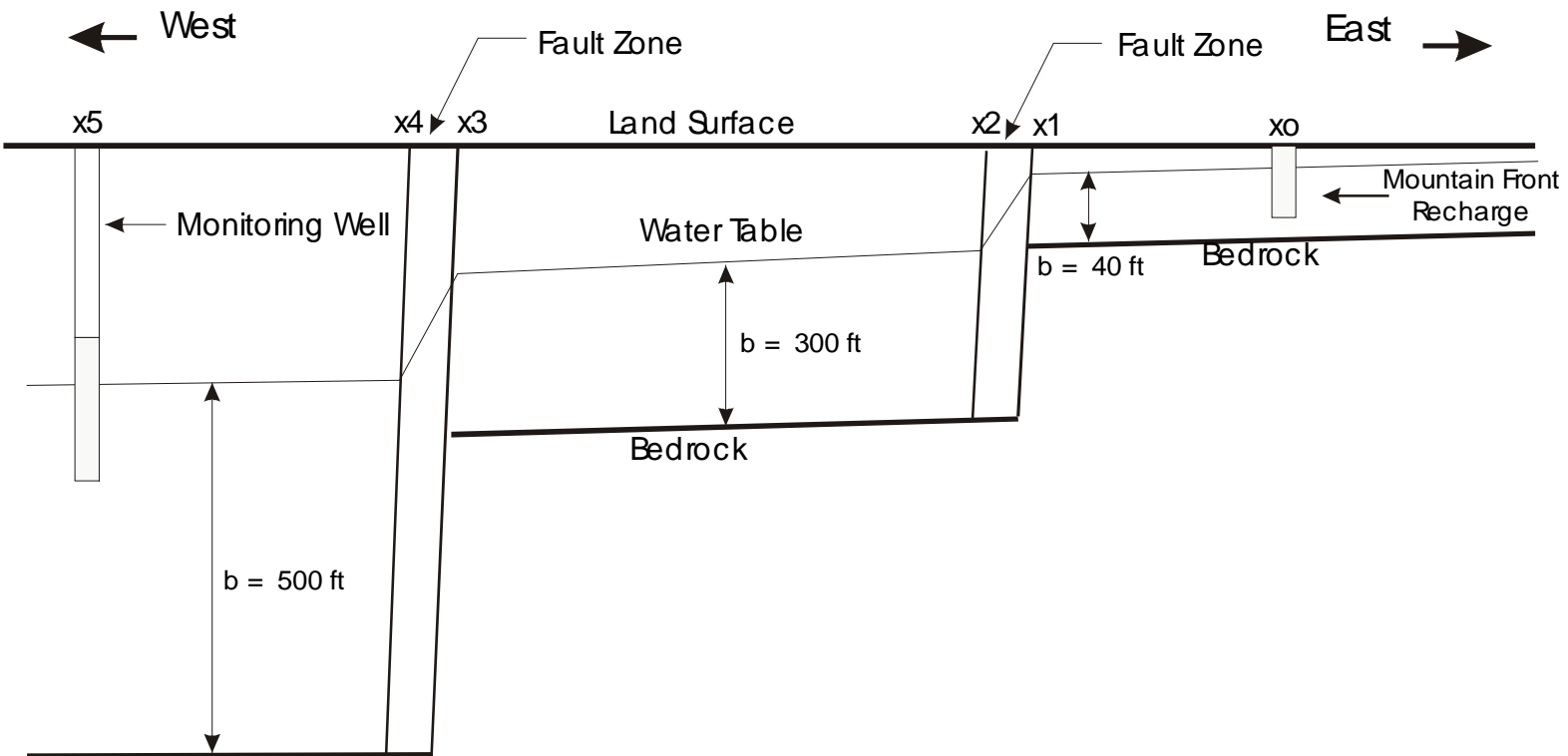
Bedrock

b = 300 ft

Bedrock

b = 500 ft

Figure 0-3. Elevation view of the conceptual model of ground water flow to the west underlying the Sandia Heights subdivision (no scale).

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In order to calculate the hydraulic conductivity of the fault zones one dimensional ground water flow is assumed. Darcy's law is used to model this flow:

$$Q = -K A \frac{dh}{dl}$$
(5-5)

where Q is the horizontal flow rate, A is the cross sectional area of the saturated formation (thickness x width), K is the hydraulic conductivity, and dh/dl is the hydraulic gradient or slope of the water table. For the Sandia Heights area, Q is assumed to be constant and its value is assumed to be equal to the mountain front recharge rate estimated by the USGS model. Using water table elevation data from the monitoring wells in the region, the hydraulic gradient can be determined along with the ground water elevation. A linear system of equations can be solved for the unknown transmissivities in each zone of the model. Values for the thickness of each zone are estimated from drilling data and from the geologic framework established by Hawley and Haase (1992). The system of equations can be set up using the head values in each zone of aquifer, reported values for the hydraulic conductivity, and the formation thickness. These further assumptions are necessary:

1. Steady state flow with Q equal to the mountain front recharge
2. The two faults have the same width and are homogeneous and isotropic within the fault zone
3. The slope of the water table is constant across each zone of the cross-section

It is recognized that this conceptual model represents a substantial simplification of the geohydrology of the region, however, there is no other existing data available which can be used to derive the values for the hydraulic conductivity in the immediate vicinity of the faults. Furthermore, it is important to recognize that the primary objective of this modeling exercise was to simulate contaminant transport in the ground water, not necessarily to develop a new model of ground water flow in the area. Simulating contaminant transport requires knowledge of the ground water flow rate and flow field (i.e. the ground water velocity vector field). How these

flow rates and directions are determined is of secondary importance provided they are representative of the conditions in the formation.

The calculations leading to the estimated values of the hydraulic conductivity are illustrated here. First estimate the flow rate per unit length along each fault (q) due to the mountain front recharge:

$$q = \frac{12.22 \text{ ft }^3/s}{10 \text{ miles x } 5{,}280 \text{ ft/mile}} = 2.31\text{x}10^{-4} \text{ ft}^2/s \qquad (5\text{-}6)$$

Next estimate the slope of the water table by rearranging Darcy's Law (equation 5-5):

$$\frac{dh}{dl} = \frac{q}{Kb} \qquad (5\text{-}7)$$

where K is the hydraulic conductivity and b is the thickness of the formation. The value of the hydraulic conductivity in the Santa Fe formation is reported to be $4.63\text{x}10^{-5}$ ft/s. Using the thickness of each zone from as illustrated in Figure 0-3 the slope of the water table in the region between the each of the fault zones can be calculated as:

$$slope_1 = \frac{2.31\text{x}10^{-4} \text{ ft}^2/s}{4.63x10^{-5} \text{ ft/s x 40 ft}} = 1.25\text{x}10^{-1} \qquad (5\text{-}8)$$

$$slope_2 = \frac{2.31\text{x}10^{-4} \text{ ft}^2/s}{4.63x10^{-5} \text{ ft/s x 300 ft}} = 1.66\text{x}10^{-2} \qquad (5\text{-}9)$$

$$slope_3 = \frac{2.31\text{x}10^{-4} \text{ ft}^2/s}{4.63x10^{-5} \text{ ft/s x 500 ft}} = 9.90\text{x}10^{-3} \qquad (5\text{-}10)$$

The slope of the water table in each of the faults zones ($slope_f$) is given by:

$$slope_f = \frac{q}{K_f b_f} \qquad (5\text{-}11)$$

The following system of linear equations can be set up to calculate the heads at chosen values of x across the model using the assumption that the slope of the water table is constant in each zone:

$$h_1 = h_0 - slope_1 (x_1 - x_0) \qquad (5\text{-}12)$$

$$h_2 = h_1 - q / K_f b_f (x_2 - x_1) \qquad (5\text{-}13)$$

$$h_3 = h_2 - slope_2 (x_3 - x_2) \qquad (5\text{-}14)$$

$$h_4 = h_3 - q / K_f b_f (x_4 - x_3) \tag{5-15}$$

$$h_5 = h_4 - slope_3 (x_5 - x_4) \tag{5-16}$$

In this system of equations x_o, and x_5 refer to the locations of the up gradient and down gradient wells, while h_o and h_5 refer to the water table elevations at these points. x_1, x_2, x_3, and x_4 represent the up gradient and down gradient locations of the faults (see Figure 0-3) and h_1, h_2, h_3, and h_4 are the corresponding water table elevations at these boundaries. The transmissivity of each fault zone is the product of the conductivity and the saturated thickness, $T_f = K_f b_f$, and w_f is the width of each fault ($w_f = (x_2 - x_1) = (x_4 - x_3)$), then equations 5-12 through 5-16 can be added and solved for T_f:

$$T_f = \frac{2qw}{(h_o - h_5) - q\left(\dfrac{(x_1 - x_o)}{T_1} + \dfrac{(x_3 - x_2)}{T_2} + \dfrac{(x_5 - x_4)}{T_3}\right)} \tag{5-17}$$

The thicknesses of each zone of 40 ft, 300 ft, and 500 ft, is multiplied by the hydraulic conductivity of 4.63×10^{-5} ft / s to get the transmissivities T_1, T_2 and T_3, and plugging the following values into equation equation 5-17:

$q = 2.31 \times 10^{-4}$ ft^2 / s

$h_0 = 6410$ ft   (head at monitoring well EG01)

$h_5 = 5606$ ft   (head at monitoring well SH03)

$x_0 = 0$ ft   (eastern boundary)

$x_1 = 1725$ ft   (distance to eastern fault)

$x_2 = 2070$ ft   (distance across the first fault)

$x_3 = 4830$ ft   (distance between faults)

$x_4 = 5175$ ft   (distance across second fault)

$x_5 = 6555$ ft   (distance to SH03, known head)

$T_1 = 0.002$ ft^2 / s

$T_2 = 0.014$ ft^2 / s

$T_3 = 0.023$ ft^2 / s

$$w = 345 \text{ ft} \qquad \text{(one cell width = fault width)}$$

yields a transmissivity for each fault zone of 0.0003 ft^2 / s. This value was used as input to the MODFLOW model to simulate the ground water flow field beneath the Sandia Heights subdivision. The water table elevations for ground water in the the Sandia Heights neighborhood is shown in Figure 0-4.

1.14.3. Model Results

The output from ground water flow model (MODFLOW) is the value of the hydrostatic head at each node, while the output from the contaminant transport model (MOC) consists of concentration values for each cell in the model. The water table elevations are shown in Figure 0-4 and the nitrate concentrations predicted by the transport model are plotted as contours in The effects of the low conductivity faults on the water table is shown by the drop of the table across the two faults in Figure 0-4. The vertical profile is similar to that illustrated in the conceptual model (Figure 0-3).

The highest nitrate concentrations in the model study area occur between the two faults which suggests that the initial distribution of the nitrates is controlled by the geometry of the faults which were modeled as low hydraulic conductivity barriers as described in the preceding section. For a 30 year simulation the highest concentration is approximately 5.5 mg/L under the northwest section of the subdivision.

It is important to note that the model did not include addition of contaminants west of Tramway Boulevard as the objective was to model the impact of unsewered development on ground water quality in Sandia Heights and the land immediately west of Tramway Boulevard is not completely developed at present. This had the effect of producing low nitrate concentrations along the western boundary of the model, in the area known as Albuquerque Acres. In actuality, on-site wastewater systems in Albuquerque Acres will contribute to ground water pollution and the water will become increasingly contaminated as it flows to the west.

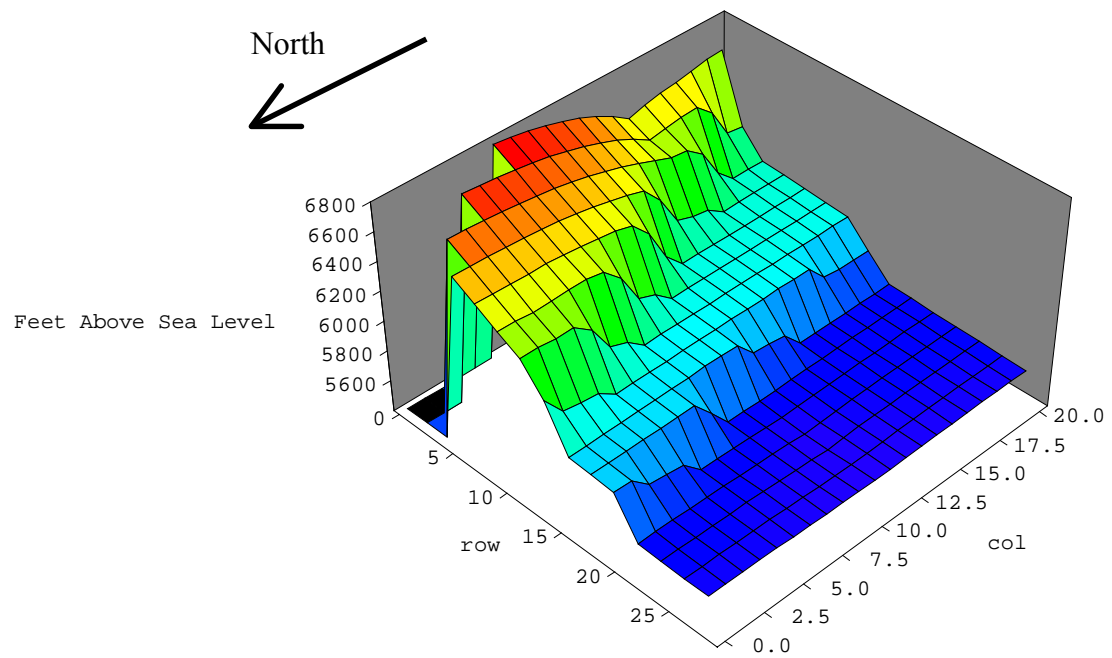Figure 0-4. Water table elevation predicted by the MOC model for Sandia Heights.

Waste-water
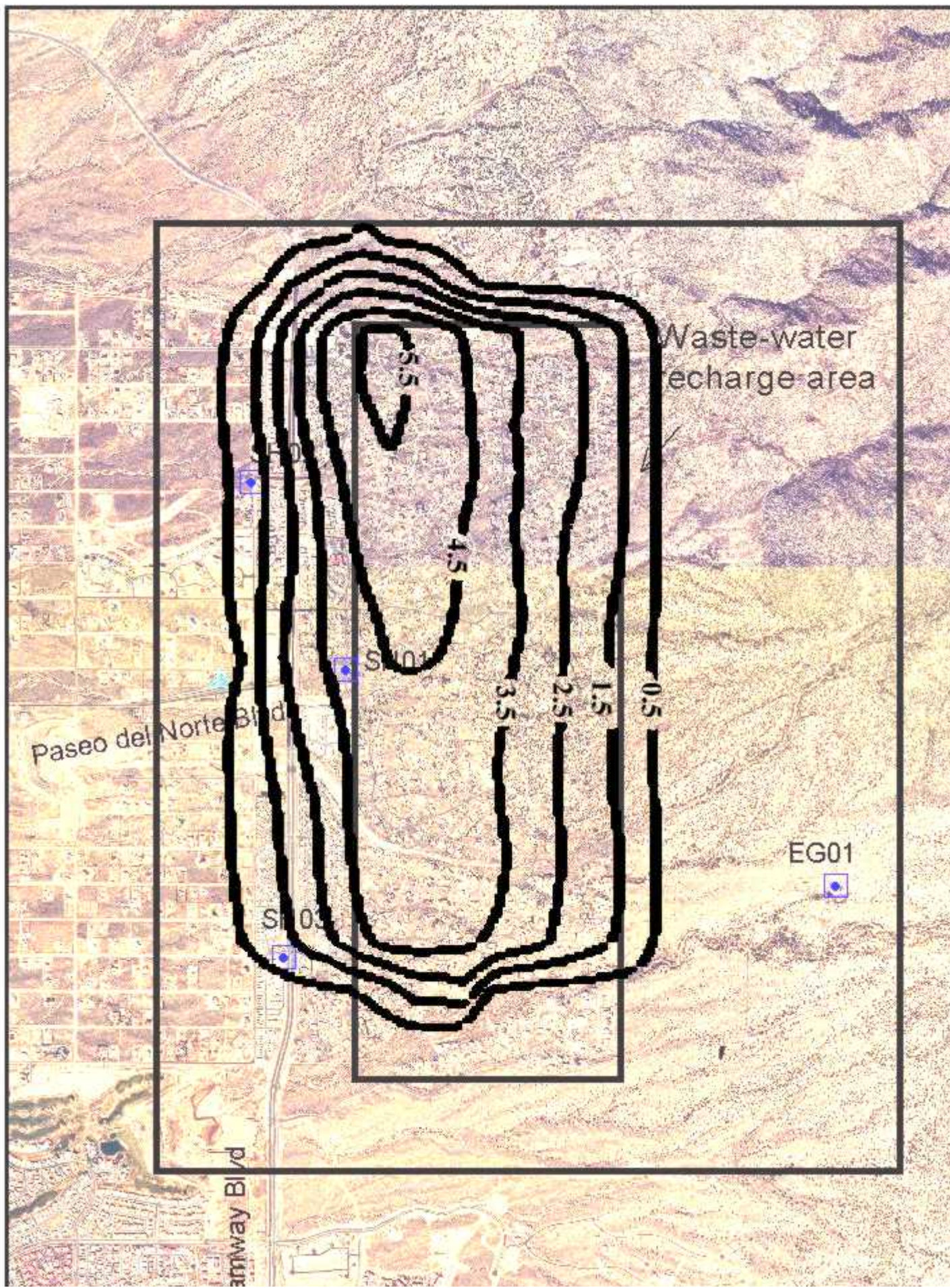recharge area

Paseo del Norte Blvd

EG01

Figure 0-5. Nitrate concentrations (mg N/L) predicted by MOC model of Sandia Heights subdivision after 50 years.

The concentrations from sampling the monitoring wells and the nitrate concentrations predicted by the model are compared in Table 0-2. There is reasonable agreement between the measured and predicted values. The model increases the up gradient boundary head by 100 feet. This is a function of the mountain front recharge rate and the low conductivity of the faults, causing the head to rise behind the faults.

	Average Nitrate Conc. (mg/L)	Predicted Nitrate Conc. (mg/L)	Measured Heads (ft)	Predicted Heads (ft)
EG01	0.12	0.0	6410(?)	6509
SH01	1.33	4.4	5603	5625
SH02	3.31	4.5	5632	5630
SH03	1.42	2.4	5606	5628

Table 0-2. Comparison of measured and predicted head and nitrate concentration values.

The model was run for time periods up to 150 years to predict when and where the nitrate concentrations might exceed the NM ground water standard of 10 mg N /L as NO_3^-. Figure 0-6 plots the nitrate concentrations along the direction of flow (from east to west) as a function of time for the model cell with the highest values. Because the model does not include injection wells to simulate recharge from on-site disposal systems beyond the western boundary of the model (i.e. west of Tramway Boulevard) the nitrate concentrations at the 30 year time frame drops back to zero. This reflects the assumption that the initial concentration of nitrate was 0 mg N/L as nitrate in the aquifer, and supports the theory that the nitrate plume from the Sandia Heights subdivision is moving west towards North Albuquerque Acres.

From Figure 0-1 and Figure 0-5 note that monitoring well SH01 is located on the western boundary of the wastewater recharge area while monitoring wells SH02 and SH03 are slightly to the west. The contours plotted in Figure 0-5 show that after 50 years of recharge these latter two monitoring wells are on the western edge of the contaminant plume and the ground water nitrate concentrations have not yet reached the maximum steady values.
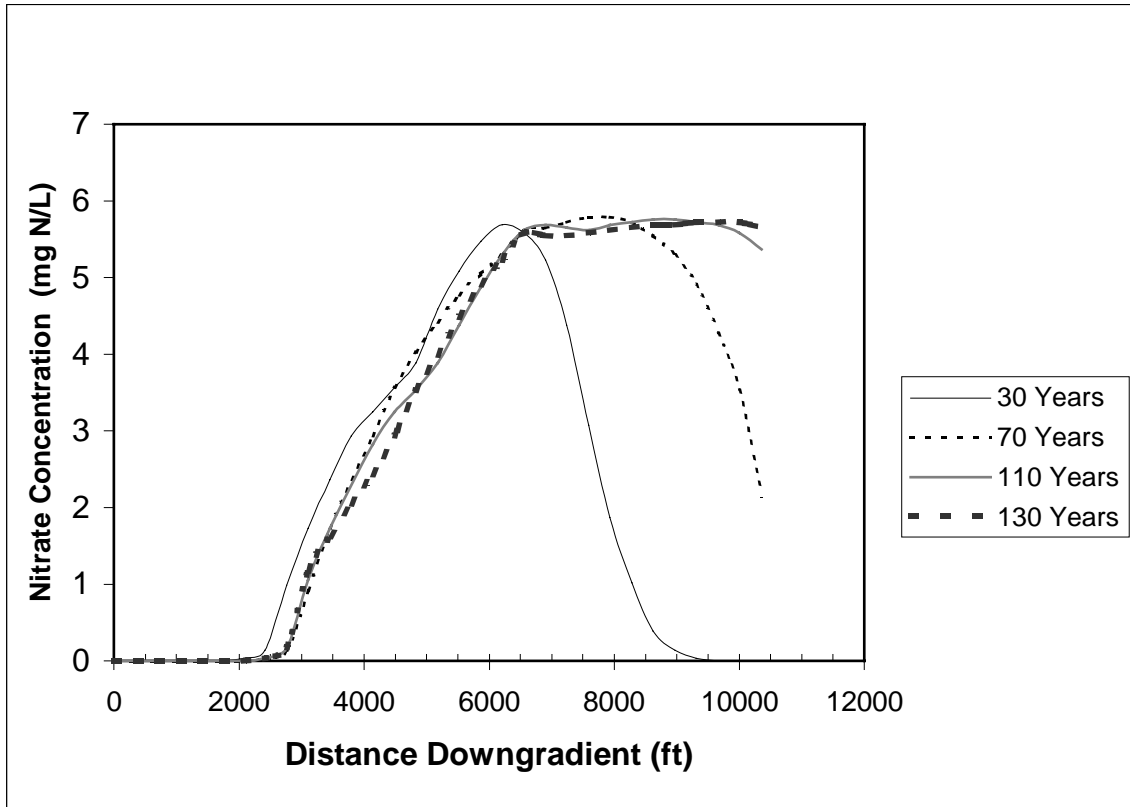
Figure 0-6. Nitrate concentration along down gradient axis as function of time.
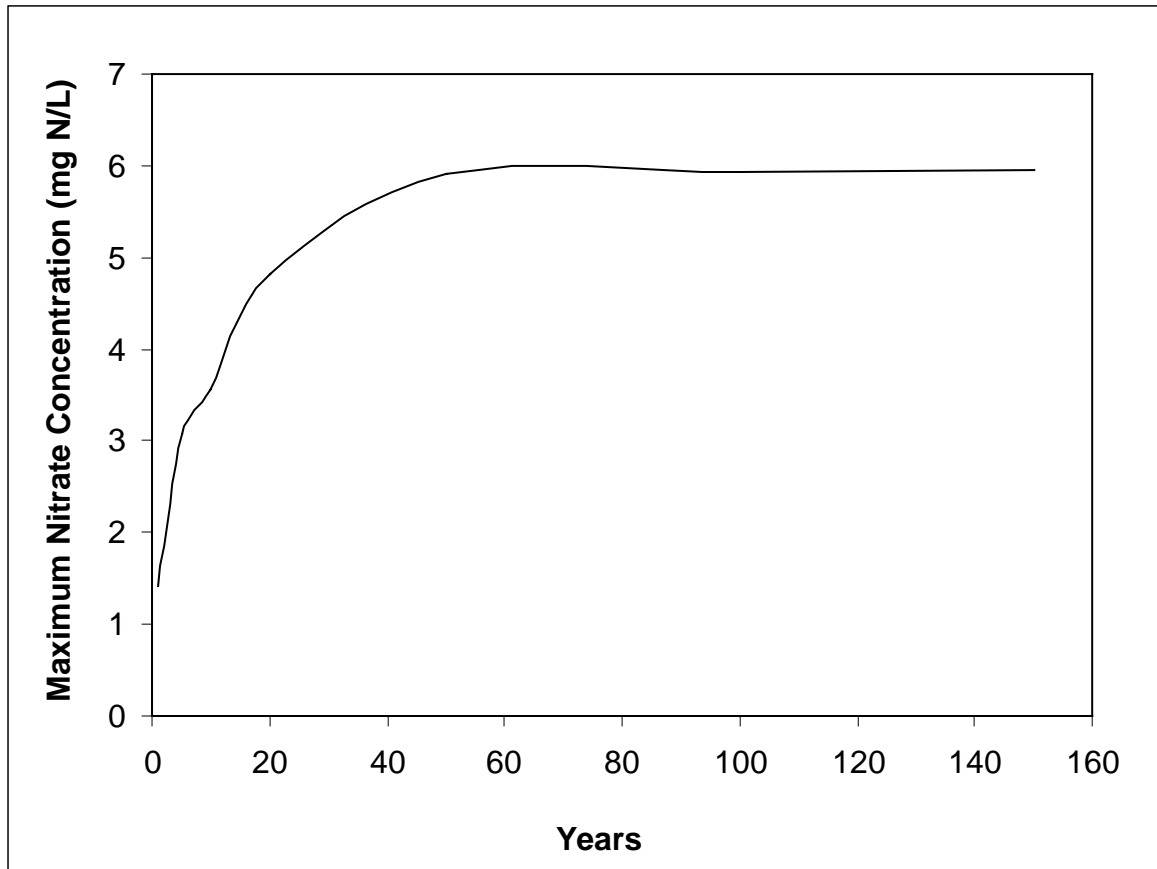
Figure 0-7. Maximum nitrate concentrations with time predicted by the Sandia Heights contaminant transport modelling.

Within the Sandia Heights study area the maximum steady state nitrate concentration of approximately 6 mg N/L is reached in approximately 50 years (Figure 0-7) and a steady state value of 6 mg N/L is reached at the down gradient boundary of the modeled area after 120 years. A rough estimate can be made of the velocity of contaminant front from Figure 0-6. The midpoint of the contaminant front is about 3 mg/L. The horizontal distance between the 30 year curve and the 70 year curve at 3 mg/L is 2100 ft. Over 40 years this is an average velocity of 53 ft/yr. Not shown on this graph is the fact that the migration velocity of the plume decreases as the plume moves to the west. This is due to the increased thickness of the saturated zone in the Albuquerque Basin west of the Tramway faults. This westward migration velocity is part of the explanation for the unexpectedly low nitrate concentrations in the ground water monitoring wells along Tramway Boulevard.

1.14.4. Parameter Variation

Porosity, septic tank density, nitrogen loading, and mountain front recharge values were varied and model runs were conducted for 1, 5, 10, 20, 50, and 100 years for each variation. A curve for each variation of the parameter under consideration was produced with the maximum nitrate concentration in the grid plotted against time. Figure 0-8 and Figure 0-9 summarize the results associated with varying each parameter individually. For each scenario it is apparent that steady state conditions are reached after approximately 50 years. Figure 0-8 shows that varying the porosity between 0.1 and 0.3 only changed the time to reach the maximum nitrate concentrations in the model, with the higher porosity achieving the maximum nitrate concentration in 20 years and the lowest porosity requiring 60 years to reach maximum values. This is explained by the fact that, for a given volumetric flow rate, the ground water velocity is inversely proportional to the porosity of the formation. Variation in lot density is shown in Figure 0-9. Using development density values ranging from one septic tank per 0.4 acres to one per 1.6 acres yields a maximum nitrate concentrations ranging from 10 mg N/L to 2.5 mg N/L for the geohydrologic conditions in this model. Remember the current average lot size of the Sandia Heights subdivision is approximately 0.8 acres/residence. Figure 0-10 shows the effects of halving and doubling the nitrogen loading. These changes predict that the highest nitrate concentration would be 12 mg N/L. The variation of nitrate concentration was linearly proportional to the total nitrogen loading in contrast to the other two parameters. To investigate the effect of changing the mountain front recharge rate on the nitrate concentrations, the recharge was varied by $\pm$ 50% from the value of 12.21 ft^3/s predicted by the USGS (1996). The results of these changes are shown in Table 0-3. These results show that increasing the mountain front recharge rate reduces the nitrate concentration, which is attributable to dilution by the greater volume of water flowing beneath the subdivision.
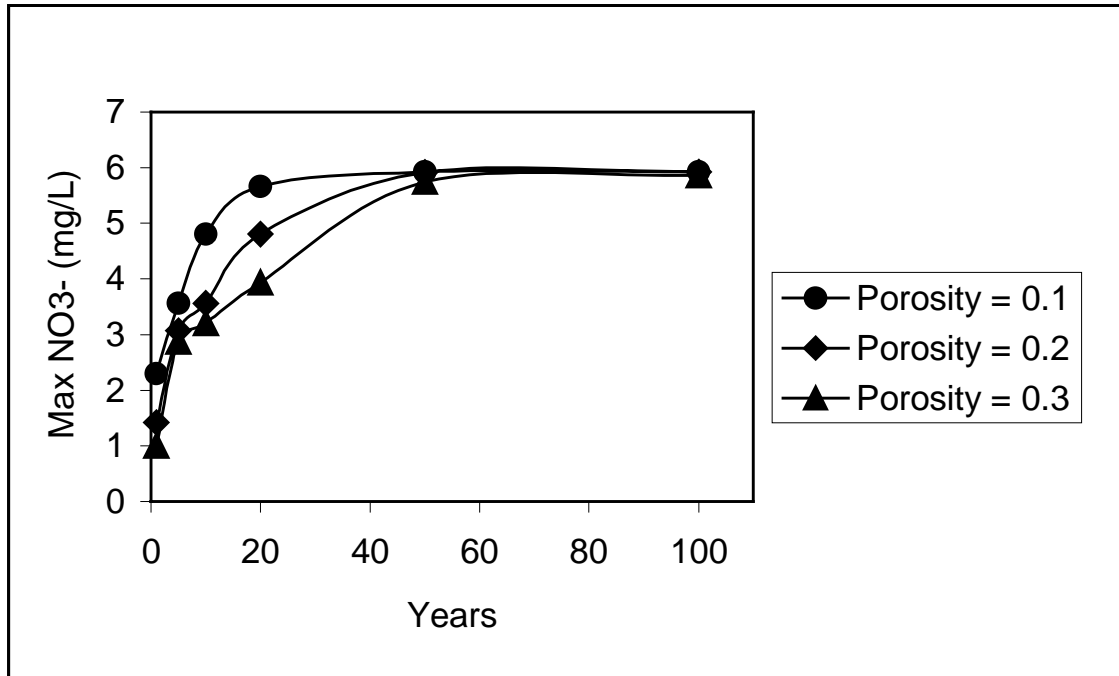
Figure 0-8. Relationship between porosity and maximum nitrate concentration.
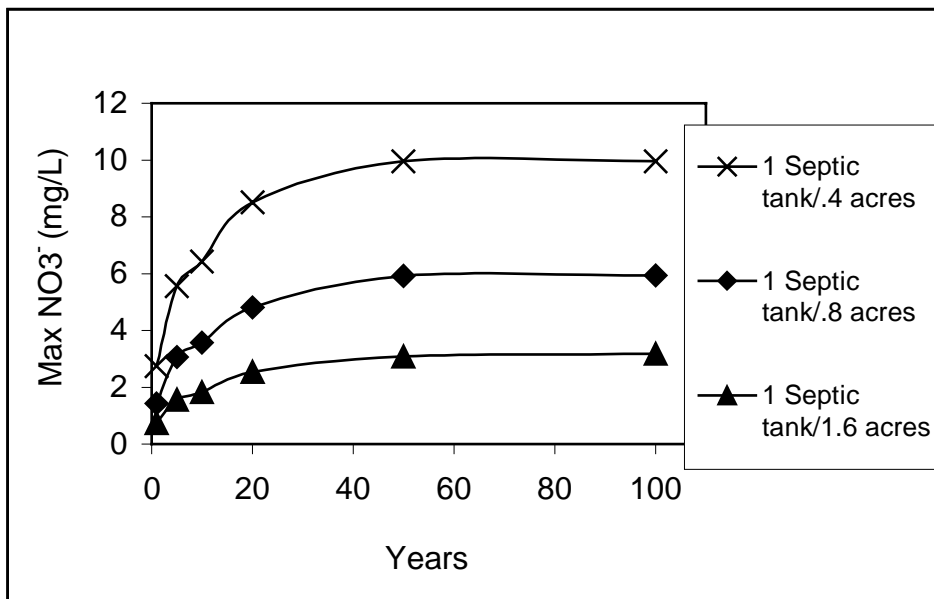
Figure 0-9. Variation in density (Lot Size) of septic tanks and predicted nitrate concentrations.
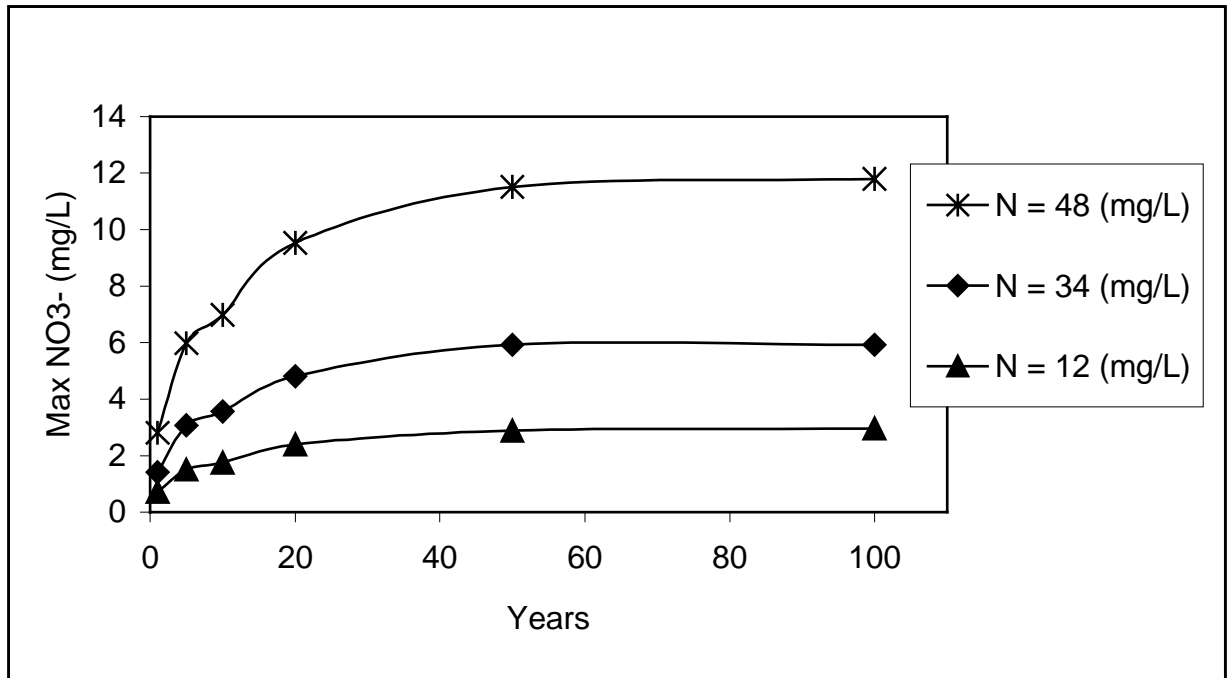
Figure 0-10. Variation of nitrogen loading with predicted nitrate concentrations.

Recharge (ft^3/s)	Maximum Predicted Nitrate Conc. (mg N/L)
6.11	7.1
12.21	5.7
18.32	4.2

Table 0-3. Variation of maximum nitrate concentration with mountain front recharge rate.

A direct comparison of the effects of the three most important parameters (lot density, leachate total nitrogen concentration, and mountain front recharge) are summarize in Figure 0-11.
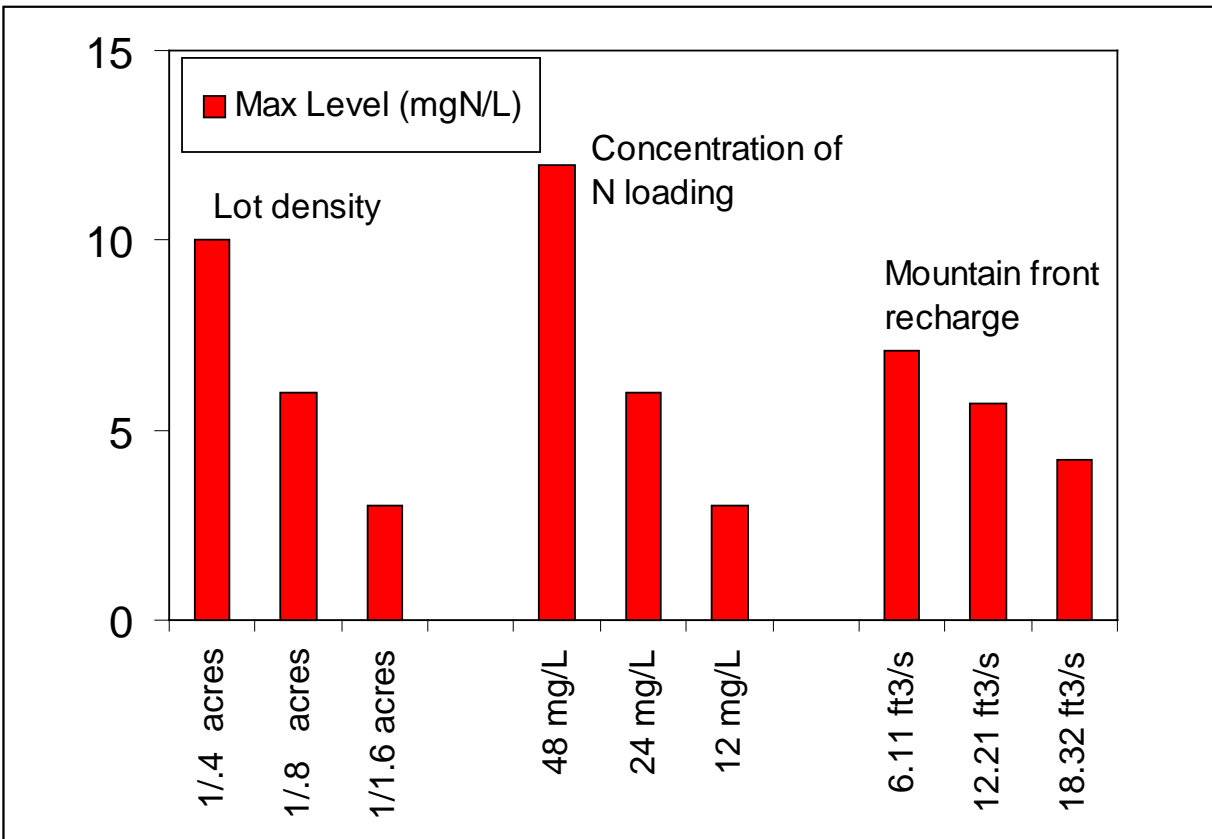
Figure 0-11. Comparison of lot density, leachate nitrogen concentration, and mountain front recharge rates on the maximum nitrate concentration in ground water beneath the Sandia Heights neighborhood.

1.14.5. Discussion of Model Results

The results of the ground water flow and contaminant transport modeling demonstrate that the three main factors that will determine if the ground water standard of 10 mg N/L as NO_3^- is exceeded are: 1) the density of on-site wastewater disposal systems, 2) the total nitrogen loading from each system, and 3) the mountain front recharge rate. In principle the first two parameters can be controlled by restrictions on lot size or type of on-site wastewater treatment and disposal system. The reason that the mountain front recharge rate was studied was to determine the effect of dilution on the maximum ground water nitrate concentrations. For the present development density of 1.25 septic tanks per acre (0.8 acres average lot size in Sandia Heights), the absorption field leachate must be less than approximately 40 mg N/L to comply with the State ground water standard of 10 mg N/L as nitrate. The model also predicts that the

nitrate contamination front is moving down gradient at a velocity ranging from 40 to 70 ft/yr. Note that in this modeling exercise, nitrogen is considered to be a non-reactive conservative tracer that moves with ground water, is not retarded by interaction with soil particles, and is not subject to biological uptake or denitrification reactions.

Comparison of the modeling results with those obtained by the field monitoring program (Chapter 4) provide qualitative confirmation of the model as well as an explanation for the unexpectedly low nitrate concentrations in the down gradient monitoring wells. First, it is important to note that the ground water elevations measured in the monitoring wells cannot be used to calibrate the flow model; these values were instead used to provide input to the flow model to estimate the hydraulic conductivity across the two faults which extend along the length of the area modeled as described in Section 5.1.3. However, the contaminant transport modeling results can be compared to the water quality of the wells. The model shows that there are two phenomena which are contributing to the low nitrate concentrations, the ground water contaminant profile may not yet be fully developed at the location of the down gradient monitoring wells and high degree of dilution is expected to occur due to the relatively large mountain front recharge rate.

The Sandia Heights neighborhood was first developed in the early 1970's; aerial photographs taken in 1968 show no houses at all, while development of all available building sites in the area has occurred only within the last 10 years. The contaminant transport modeling shows that the time required for a fully developed contaminant front to reach the down gradient monitoring wells is between 30 and 50 years. Thus, one interpretation of the results from the monitoring wells is that they represent early values of a contaminant plume migrating to the west, and that the nitrate concentrations will continue to increase with time. If, as the model predicts, the contaminant front is expected to take 30 years to develop fully and the maximum ground water nitrate concentration will be 6 mg/L, these results suggest that the nitrate concentrations in the monitoring wells will continue to increase over the next 15 years. As discussed in Section 5.1.2 the contaminant transport model does not consider transport of nitrogen species through the vadose zone or loss of nitrogen to plant uptake or microbial denitrification. Furthermore, this model assumes an instantaneous input of the full nitrogen loading rate from on-site disposal systems, rather than a gradual increase in loading as the community is developed over time.

Including more realistic assumptions regarding the rate of development of the subdivision and contaminant transformation and transport through the vadose zone will result in prediction of later development of a contaminant plume than predicted by this modeling study. Further discussion of nitrate transport and reactions in the vadose zone are presented in sections 4.3 and 5.2.

The second factor that is believed to contribute to the unexpectedly low nitrate concentrations in the monitoring wells is dilution by the high ground water flow rate beneath the area. In the Sandia Heights area the ground water flow rate is principally determined by the mountain front recharge. Reducing or increasing the recharge rate by 50% (Table 0-3) results in a corresponding 25% increase or decrease in the maximum nitrate concentrations, with the lower recharge rates producing greater nitrate concentrations. This result of the modeling exercise means that high density development in areas with lower recharge rates such as North Albuquerque Acres will be more vulnerable to ground water contamination from on site disposal systems.

1.15. Modeling of Vadose Zone Transport

1.15.1. Simple travel time estimates in vadose zone

A simple approach to estimating the time it takes for the septic tank effluent to reach ground water is to use the concept of field capacity. The field capacity is the water content at which gravity drainage becomes negligible. Soils beneath the leach field will increase in water content from the ambient value to the field capacity before effluent moves downward to lower depths. The amount of water (Q) that has to be added to the soil before effluent finally reaches ground water is

$$Q = (\Theta_{fc} - \Theta_a) * V \qquad (5\text{-}2)$$

where V is the volume of soil between the water table and the leach field that becomes wetted, Θ_{fc} is the field capacity, and Θ_a is the ambient water content of the soils. If the rate at which water is added to the soil through the leach field (q) is known, then the time (t) for the effluent to reach ground water is given by

$$t = \frac{(\Theta_{fc} - \Theta_a) * V}{q} \qquad (5\text{-}3)$$

There are many different possible flow geometries which correspond to different volumes of soil that will be wetted due to wastewater flow from the leach field. The simplest flow geometry would be the case of only vertical, downward flow. In this case, for a q of 150 ft^3/day (estimated from a metering valve on the water supply to the Sheriff's substation) distributed over an area of 2500 ft^2, a depth to ground water of 500 ft, an average field capacity of 14% (volumetric basis), and no ambient water, the time for effluent to reach the ground water will be about 3 years.

Even in the coarse soils that were found beneath the leach field, some lateral spreading of the wetting front from the leach field is expected. If it is assumed that the effluent spreads out 1

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ft horizontally for every 2 ft it moves downward, the affected volume of soil will increase greatly by the time the ground water table is reached. In this scenario, the time required to reach the ground water table will be about 35 years.

A geometry that would constitute an upward bound of lateral spreading would be if the volume of wetted soil resembled a hemisphere by the time it reached the ground water table. With this geometry, the effluent would spread laterally in every direction an amount equal to the distance it moved downward. The time required to reach ground water is about 500 years with this flow geometry assumption.

The simple approach described above represents a gross simplification of a complicated problem. This approach is a water balance approach, and does not account for the mechanisms and factors that control flow. Further, it ignores the influence the heterogenities in a thick vadose zone will have on the flow. However, the numbers are useful as bounding estimates of travel time.

1.15.2. Numerical simulations

1.15.2.1. Introduction

The transport and transformation of septic tank effluent through a vadose zone to underlying ground water is a very complicated process. Complete modeling of this process would include three dimensional, transient transport (multiphase flow) along with coincident geochemical modeling of nitrogen transformations. Some fundamental information required to such modeling efforts have large uncertainties associated with them, including

1. effluent composition, rate quantity, and areal distribution within the leach field,

2. vadose zone lithology, properties, and initial conditons, and

3. nitrogen to nitrate conversion ratios in the vadose zone.

Consequently, the modeling presented here focused on estimating travel time of infiltrating water (e.g., septic tank effluent) through the vadose zone.   Travel time estimates are very useful to provide a frame of reference for considering septic tank effluent movement and possible contamination of ground water supplies. Geochemical transformations were not included in any modeling results. Future work can focus on developing more complete hydrologic and geochemical models.

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Two numerical models were created. The first model approximated the vadose zone beneath the leach field at the Sheriff's substation that was the subject of the vadose zone investigation described in Chapter 3. The purpose of simulations using this model was to assess the reasonable of model results (travel times) when compared to the experimental results. This model extended from the ground surface to a depth of 70 feet. The second model extended the vadose zone to 500 feet. These simulations were designed to obtain bounding estimates of travel time from the leach field to ground water.

The computer program used to conduct these simulations was the TRACR3D computer code (Travis and Birdsell, 1991). This code solves the equations of transient two-phase flow and multicomponent transport in deformable, heterogeneous, sorptive, porous media. The code solves these equations using an implicit finite difference scheme for flow and implicit approach for transport. The user's manual (Travis and Birsell, 1991) provides a complete model description as well as a guide to code operation and options.

1.15.2.2.Model of leach field

The modeled geometry consisted of a 2-dimensional grid of 80 ft deep by 33 ft wide. There were 30 cells or elements in the vertical direction, and 10 cells in the horizontal direction. Each cell was assumed to be 3.3 ft thick in the out-of-plane direction. The left-most vertical boundary represents a line of symmetry that splits the leach field in half. The vertical boundaries are no-flow boundaries. A constant flux is applied to first cell (upper, left corner) to simulate input into the vadose zone from the distribution pipe in the leach field. The flux rate was calculated based on a constant flow of 500 gallons per day distributed over a 100 ft length within the distribution pipe.

The simulations were conducted with two soil types: a silty sand and a well-graded sand. Consistent with the Unified Soil Classification System, the silty sand is designated SM and the well-graded sand is designated SW. The hydrologic properties of these soils were described using van Genuchten's functions. The parameters used in the simulations were obtained from Carsel and Parrish (1988) and are given in Table 5.4.

Material Type	Θ_r	Θ_s	α (cm^{-1})	n	K_s (cm/s)
GW	0.03	0.4	0.212	3.43	0.5
GM	0.03	0.42	0.294	3.28	0.1
SW	0.045	0.43	0.145	2.68	0.00825
SM	0.067	0.45	0.020	1.41	0.000125

Table 0-4. van Genuchten parameters used in the SWMS-2D code.

The properties of these soils are within the range of the properties of many of the soils sampled from beneath the leach field (Table 4-6), particularly the soils with lower hydraulic conductivities.

Simulations were conducted with three different assumed soil profiles: (1) homogeneous SM, (2) homogeneous SW, and (3) SW with a 5 ft thick SM layer located at a depth of 13 ft below the ground surface. The original initial intent was to model the lithology beneath the Sheriff's substation. However, the additional complexity and uncertainty introduced by including numerous discrete layers was considered to outweigh the value of the results. The initial condition of the soils was set to a low volumetric water content of 4 to 12%, consistent with the water contents measured in the background holes near the leach field (Table 4-7).

The soil was assumed to be wetted by the infiltrating water when the volumetric water content increased by 1%. Arrival of the wetting front at a depth of 65 feet from the constant rate infiltration is given in Table 0-5 for the three soil profiles simulated.

SW model (homogeneous)	SM model (homogeneous)	SW with 5 ft SM layer
45 days	1650 days	210 days

Table 0-5. Simulated time to reach 65 foot depth

The numerical results from both the SW model and the SW with SM layer model are consistent with the experimental result that effluent had passed the 65 foot depth within a year.

These results indicate that the downward movement of infiltrating water is a function of the soil properties. The SW model indicates that this type of soil will permit rapid downward movement of water. Soils in the profile with similar properties, or soils with a greater

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conductivity, will not provide significant resistance to rapid movement of septic tank effluent toward ground water. At the same time, the results of the SW with SM model indicate that a relatively thin layer of a lower conductivity soil can significantly increase travel time. The results with the SM model indicates that a homogeneous model comprised of the lower conductivity material provides a upper bound of travel time for a soil profile that includes SM layers.

1.15.2.3.Model to ground water

The second model type varied from the leach field models in its dimensions. The model extended from the ground surface to a depth of 500 feet, the assumed distance to the ground water surface. The width of the model was 333 feet. There were 30 cells in the vertical direction and 10 cells in the horizontal direction. The cells were 3.33 ft thick in the out-of-plane direction.

Three soil profiles were assumed with this model: homogeneous SW, homogeneous SM, and SW with a 33 ft thick layer of SM soil at a depth of 100 ft below the ground surface. The soil properties are given in Table 5.4. Results from these simulations are given in Table 0-6.

SW model (homogeneous)	SM model (homogeneous)	SW with 33 ft SM layer
4.5 years	126 years	20.2 years

Table 0-6. Simulated time to reach 500 foot depth (ground water).

Many of the same observations that were made from the leach field model can be made from these results. For example, the effect of a layer of a lower conductivity material within the soil profile is apparent from these results.

It is interesting to compare these results to the simple estimates of travel time through a 500 foot thick vadose zone made in 5.2.1. These results indicate a travel time between 3 and 500 years, depending on the assumed flow geometry. These results are consistent with the simulation results.

CONCLUSIONS

1.16. Ground Water Monitoring Program

The principal objective of the ground water monitoring program was to document changes in ground water quality due to on-site wastewater treatment and disposal systems. A system of six monitoring wells was installed and two additional wells were identified which were constructed and located so that they also could be used as monitoring wells. This network of monitoring wells was sampled over the course of one year and the water samples were analyzed for constituents which would indicate the presence of ground water pollution as well as could be used to interpret the geochemistry of the formation. The ground water systems sampled in this study are deeper than those which have been the subject of most previous investigations. Furthermore, all previous studies of regional ground water contamination in Bernalillo County have been conducted using water samples from domestic wells. These wells are generally less appropriate for monitoring ground water quality because:

- They usually are not screened across the top of the water table to allow sampling of water most likely to be influenced by the on-site disposal system.

- Domestic wells are constructed to produce high volumes of water, therefore, the well screen used is usually much longer than is suitable for a monitoring well.

- Many domestic wells, particularly older wells, were not constructed to limit flow down the annular space between the casing and the bore hole thus water samples from the well may be influenced by water from shallower formations.

- Because the wells were not designed and installed for the purpose of collecting ground water samples there is incomplete knowledge of the formation from which water is being pumped.

- Well construction details are frequently incomplete so that the length and depth of the screened interval is not known.

The ground water monitoring network was established in two areas of Bernalillo, four wells were located in the Sandia Heights subdivision on the western side of the Sandia Mountains, and four wells were located in heavily developed regions of the East Mountain areas

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in the Piñon Ridge, Sierra Vista, and Sandia Knolls subdivisions. A fifth well was drilled on the western boundary of the Sandia Knolls subdivision, however, no ground water was encountered and the hole was not completed. None of the areas in this study are served by sewer systems.

Based on previous work performed to identify ground water contamination in Bernalillo County and elsewhere, the primary contaminant that was expected to be present was nitrate ions (NO_3^-) resulting from biological oxidation of organic nitrogen and ammonium (NH_4^+) in the wastewater. All of the down gradient wells sampled in this study had elevated nitrate concentrations, although none were greater than 4 mg N/L. Based on the geology, the drilling records and the fairly complete chemical analyses of each sample it appears that there are two very different subsurface environments in the Sandia Heights and East Mountain study areas which at least partly explains these results.

1.16.1. Ground Water Contamination in the Sandia Heights Study Area

The geology of the Sandia Heights study area is dominated by unconsolidated fine-grained to coarse sands containing very little fractions of silt and clay sized material. The sand is mostly decomposed granite outwash from the Sandia mountains and therefore has little chemical interaction with the ground water. Depth to ground water ranges from less than 50 ft at the up gradient well to nearly 500 ft at the three down gradient monitoring wells. Due to the coarseness of the soil it is expected to be a well aerated, and this is confirmed by the presence of dissolved oxygen in all of the water samples from the monitoring wells. Nitrate is present at elevated concentrations in all of the monitoring wells providing further evidence of oxidizing conditions. There may be a slight correlation between nitrate and chloride concentrations, however, the amount that each constituent elevated is very small and there are only four points, so this correlation cannot be claimed with much confidence. Over the four quarters of ground water sampling a possible trend of increasing nitrate concentration in the three down gradient wells was noted, however, as with the chloride-nitrate correlation, there are two few data points to establish this trend with confidence. There were no other indicators of anthropogenic contamination detected in the down gradient monitoring wells in the Sandia Heights subdivision, although the samples were analyzed by the City of Albuquerque water quality laboratory for coliform bacteria and by the USGS for a suite of 67 organic compounds. These results suggest that, except for the

elevated nitrate levels, the ground water has not yet been significantly impacted by wastewater from on-site disposal systems.

In order to develop a better understanding of the ground water hydrology of the Sandia Heights subdivision, a ground water flow and contaminant transport modeling effort was undertaken. This model began with development of a conceptual model of the ground water system which incorporated the two north-south faults through the area. A ground water flow model was then calibrated to the water surface elevations in the monitoring wells. The faults were modeled as zones of low hydraulic conductivity and also serve as boundaries where the thickness of the water bearing strata increase as water flows to the west. Hydraulic parameters developed for the USGS model of the Rio Grande Basin (Kernodle et al., 1995) were used, including the mountain front recharge rate. The flow regime was then used to develop a contaminant transport model that would predict transport of nitrate in ground water undelying the Sandia Heights subdivision. The impact of on-site disposal systems was simulated by adding an injection well in each grid cell of the model which added a non-reactive conservative tracer to simulate nitrate addition to the subsurface formation by water infiltrating from leach fields.

The ground water contaminant transport model suggests that full development of a nitrate plume beneath the Sandia Heights subdivision will take 30 to 50 years due to the relatively low ground water velocity and long distances. Development in this subdivision began in the early 1970's and has only been essentially complete in about the past 10 years. The maximum concentration that is expected in this study area is approximately 6 mg N/L as nitrate. Although this subdivision is relatively densely developed at an average of 1.25 houses/acre, the high ground water recharge rate from the mountain recharge will dilute the nitrate to limit the maximum concentrations. Note, however, that as the ground water continues flowing to the west beneath the unsewered North Albuquerque Acres development, nitrate concentrations will be expected to increase. The down gradient boundary of the contaminant transport model was located along Tramway Boulevard and did not include modeling of the impact of on-site disposal systems to the west.

Thus, both the field evidence and the modeling study indicate that the ground water beneath the Sandia Heights subdivision has definitely been impacted by discharge of domestic wastewater from on-site disposal systems. The level of contamination is small at present and

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appears to be limited to nitrates produced by oxidation of reduced nitrogen species (organic nitrogen and ammonium). The modeling study suggests that the nitrate concentrations are rising and may continue to do so for another 10 to 30 years. The field data from one year of sampling appears to show increasing nitrate concentrations in water from the monitoring wells, although there are too few data points over too short a time period to confirm this trend.

1.16.2. Ground Water Contamination in the East Mountain Study Area

The hydrogeologic conditions of the East Mountain area are very different from those on the west side of the Sandia Mountains. The geologic formations are dominated by consolidated limestone, sandstone and shale formations which sharply contrast with the unconsolidated decomposed granite sands of the Sandia Heights subdivision. Ground water flow through fractures in the tight formations of this area is much more prevalent than simple Darcian flow through porous media. The ground water hydrology reflects these differences as well in that instead of their being a single unconfined regional aquifer of large lateral extent, the ground water formations are relatively limited, to a large degree they follow the surface topography, and in many locations there are multiple water bearing formations. The water chemistry of these formations in part reflects the geochemistry of the host rock, and there can be dramatic differences in the water chemistry of ground water from wells located only a few miles apart.

As part of the background investigation done in preparation for drilling monitoring wells, a detailed map of water table elevations in the area was constructed using information from the records of approximately 250 wells maintained by the Office of the State Engineer in Albuquerque, NM. Using information from this map, each of the monitoring wells was located in areas expected to exhibit the greatest evidence of contamination from on-site wastewater treatment and disposal systems. Monitoring wells were installed in the Piñon Ridge, Sierra Vista and Sandia Knolls subdivisions, and a hole was drilled in the Sandia Knolls subdivision that did not encounter ground water. Water from these monitoring wells found multiple indications of contamination from on-site disposal systems including elevated nitrate concentrations in down gradient wells.

Other indicators of ground water contamination include the presence of anthropogenic organic chemicals in all of the ground waters sampled for these constituents by the USGS, the

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presence of coliform bacteria in some of the ground water samples (although these results may be due to contamination from drilling or sampling activities), the presence of low but elevated nitrate concentrations in all of the wells, and evidence that most of the monitoring wells produce samples from formations which are under strongly reducing conditions. Thus, the explanation of the low nitrate concentrations is that much of the ground water in the East Mountain area is strongly reducing. The possible factors which contribute to this phenomenon include:

- The biochemical oxygen demand (BOD) from the wastewater associated with the high density of on-site disposal systems exceeds the oxygen transfer capacity of the soil, thus oxidation of organic nitrogen and ammonium (i.e. nitrification) does not occur. Nitrate is therefore not generated in the soil column beneath absorption fields.
- The soils in each of the subdivisions where monitoring wells are located have much higher fractions of fine grained material which will reduce oxygen transport through them.
- There may be more natural organic material in the East Mountain area soils than in the Sandia Heights study area which can serve as a substrate for microbial denitrification.
- The mineralogy of the East Mountain area soils may contain sufficient concentrations of pyrite or other reduced minerals to support microbial denitrification by chemoautotrophic bacteria (lithotrophs).

These reducing conditions either prevent the formation of nitrates or reduce nitrate concentrations in ground water or soil pore through microbial denitrification. The most important point is that there is significant evidence that ground water contamination has occurred as a result of on-site wastewater treatment and disposal systems, although nitrate concentrations currently meet State ground water standards. This evidence includes low dissolved oxygen levels, elevated concentrations of total organic carbon, detection of coliform organisms in ground water, and detection of traces of anthropogenic organic compounds in several of the East Mountain area wells.

1.17. Vadose Zone Investigations

A vadose zone investigation was conducted just west of the Sandia Heights study area in the unconsolidated sand deposits of the North Albuquerque Acres subdivision. The purpose of this study was to collect information which could be used to develop a better understanding of the vertical rate of flow of water through the unsaturated zone beneath an absorption field, and also to characterize the chemical transformations and contaminant transport from wastewater discharged to the soil from an absorption. The site selected consisted of the Bernalillo County

Sheriff's substation along Paseo del Norte road. This substation was approximately 14 months old at the time of the study. Depth to ground water at this site was believed to be greater than 500 feet.

Five bore holes were drilled to a depth of 65 feet each and samples were collected at five foot intervals. The samples were subjected to a suite of physical and hydraulic testing, and a fraction of each sample was also leached with deionized water. The leachate was then analyzed for chemical constituents to determine the solution chemistry of the pore water. The soil samples were dominated by sands with low contents of silt and clay size material in some of the samples. There was a high degree of variability of the soil characteristics between the bore holes and also at different depths within the bore holes. As expected for well drained sandy soils in an arid environment, the gravimetric moisture content in the background soil samples was very low and ranged from 2 to 9%. Soils from the bore hole most directly influenced by infiltrating wastewater from the absorption field ranged from about 10 to 15%. The physical characteristics of the soils, their hydraulic properties, and the low water contents all suggest that the vertical flow of water through the unsaturated soils at this site is generally rapid.

A bromide tracer was added to the substation wastewater two weeks before the soil sampling program. Deionized water leachate from the soil samples had low but detectable bromide concentrations in all of the samples from the hole closest to the absorption field, and no bromide concentrations in soil samples from the background bore holes. This confirms the very high vertical migration velocity of water through the unsaturated zone, at least in the top 65 ft of the soil column sampled in this study. Numerical modeling shows that the vertical migration rate is highly dependent on the presence of fine grained, low permeability soil horizons. Bounding calculations suggest that between 4.5 and 126 years may be required for leachate from an absorption field to travel vertically to the water table. Development of a better estimate of this travel time is not possible without knowledge of the soil properties throughout the vertical extent of the soil column.

The chemical analyses of leachate from the soil samples found very low concentrations of ammonium and high concentrations of nitrate in pore water from near the absorption field. This confirms in part the conceptual model developed for this project which postulates that ammonium (NH_4^+) is retained on negatively charged soil particles and will not migrate to

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significant depths beneath the absorption field system. Furthermore, in the well aerated sandy soils this ammonium is readily oxidized to nitrate (NO_3^-) which is not retained by the soils and migrates vertically with the unsaturated pore water. The chemical analyses also found that over 50% of the organic carbon in the pore water from near the absorption field was removed in the top 30 feet of soil, however, a significant fraction of this material (~20%) was transported to depths of at least 65 feet. The results show that there is adequate organic carbon to serve as a substrate for microbial denitrification if oxygen is not present. Trends for both nitrate and organic carbon species in the bore holes suggest decreasing concentrations of both compounds with depth, although there is enough scatter in the data that these trends cannot be shown statistically. One possible explanation for these trends is that microenvironments exist in the soil pore water in which microbial activity has depleted the oxygen content within the solution to the point that denitrification will occur. While these results are suggestive, based on the limited data and limited depth of the bore holes, they must be considered very preliminary. One of the principal values of the results of the deep vadose zone soil sampling study is that they suggest an important area where future research is needed.

With respect to potential impacts of on-site systems on ground water the most significant conclusions that can be drawn from the vadose zone sampling program are: 1) that water infiltration rates through the unsaturated soils on the western slope of the Sandia Mountains is rapid, and 2) oxidation of reduced forms of nitrogen to nitrate occurs in the first 10 to 20 feet of the soil column. Nitrate then migrates rapidly downward through the unsaturated zone with the vertical flow of pore water. From a ground water protection perspective, a conservative assumption should be made that all nitrate in pore water beneath the root zone will eventually reach the water table and result in a nitrate plume in ground water beneath the absorption field. Further work on possible in-situ denitrification processes in the vadose zone is needed to justify a less protective management approach.

In considering the results of the vadose zone sampling program it is important to recognize that samples were only collected in the top 65 feet of an approximately 500 foot thick unsaturated zone. This is an important limitation on the knowledge regarding contaminant transport and transformations in deeper formations.

1.18. Recommendations

As with many investigations, this one generated as many questions as it answered. Thus while it is apparent that ground water contamination has occurred as a result of on-site wastewater treatment and disposal, the degree of contamination that can be expected in a particular hydrogeologic setting can not be predicted with confidence. Furthermore, this study showed that there may be geologic conditions which can limit the maximum concentration of some pollutants, although others may be present at sufficient concentrations to pose a threat to health and the environment. Finally, this study suggests that contaminant concentrations in the Sandia Heights study area may be increasing with time which may pose a threat to down gradient water supply wells. Accordingly, the following recommendations are made.

- Continue sampling the monitoring well network established in this study on a regular basis to detect trends in ground water quality and water table elevation.
- Extend the ground water flow and contaminant transport model for the Sandia Heights subdivision to include the impact of unsewered development in North Albuquerque Acres on underlying ground water quality.
- Re-sample the monitoring wells for the presence of anthropogenic organic compounds biannually.
- Perform isotopic analyses on ground water samples from the monitoring well network to determine age of the water and source of nitrate contamination.
- Initiate development and compilation of data suitable for inclusion in a geographic information system to collect and process data regarding ground water quality throughout Bernalillo County.
- Support a formal investigation of contaminant transport and transformations in thick unsaturated formations.
- Collect water quality information relevant to on-site wastewater disposal during other studies of ground water contamination such as RCRA, Superfund, and underground storage tank investigations.
- Expand the monitoring well network to include additional wells.

These recommendations are briefly discussed in the following sections.

1.18.1. Continue Sampling Monitoring Well Network

This project established a network of 8 monitoring wells solely for the purpose of measuring the impact of on-site wastewater treatment and disposal on ground water quality. The wells were located in areas believed to be especially vulnerable to these impacts, and in all of the areas studied, these wells are the only wells designed solely for the purpose of collecting ground water samples representative of conditions at the top of the water table where contamination will first occur. With one exception (the second well installed in Sandia Park) they have been subjected to over one year of sampling and analysis, which has established a water level and water quality baseline which represents hydrogeochemical conditions in the underlying ground water in 1998. The ground water quality on both sides of the Sandia Mountains is almost certainly changing as new subdivisions are developed. Furthermore, increasing stress on ground water resources will cause a change in potentiometric elevations, a phenomenon almost certainly already noted in the Mhoon well in the Sandia Heights subdivision. Accordingly, it is recommended that the network of monitoring wells developed for this project should continue to be monitored for at least 5 more years or until the water elevation in the wells drops below the elevation of the well screen. The results presented in Chapter 4 suggest there is very little seasonal variability in ground water quality for all of the wells except the first Sandia Park well (SP01), hence it is suggested that semi-annual sampling be conducted. The same chemical analyses as presented in Table 3-5 should be performed, and water elevation measurements should be taken. It is also recommended that all chemical analyses be performed by the City of Albuquerque Water Quality Laboratory to maintain the maximum consistency in the data.

1.18.2. Ground Water Flow and Contaminant Transport Modeling

The ground water flow and contaminant transport modeling described in Chapter 5 was limited to the Sandia Heights area. However, the results showed a substantial impact of this subdivision on ground water quality towards the west. One of the limitations of the modeling effort performed in this study was the lack of sufficient data to develop an ab-initio model of ground water flow in this region. The presence of two faults in the area was particularly difficult

to incorporate in this model, and the value of the mountain front recharge rate was based on an average value obtained over a length of 10 miles. Better field data would improve the model. A further limitation was incomplete knowledge of the thickness of each formation appropriate for the model. Finally, and most importantly, the model did not include the impacts of additional unsewered residential development in North Albuquerque Acres, which is beyond the western, down gradient boundary of the modeled area. Due to the size of development in this area, the high density of legally platted lots in the region, and the likelihood that nitrate contamination will increase with time, a recommendation is made that the model developed in this study be refined and extended to include the entire area encompassed by North Albuquerque Acres. Since the first step in contaminant transport modeling is development of a flow model, it may be possible to use the flow component from the USGS ground water model of the Albuquerque Basin. However, this model needs to be examined in the context of the requirements and compatibility for a contaminant transport model. This includes use of a grid size appropriate for the transport model, selection of the proper aquifer thickness, and confidence that the model captures all of the physical attributes of the area that are relevant to contaminant transport modeling (for example fault zones).

1.18.3. Re-Sample Monitoring Wells for Organic Compounds

Throughout the country, most of the emphasis on ground water contamination from on-site systems has been on the appearance of elevated nitrate concentrations, principally because it is the one drinking water contaminant that commonly appears as a result of on-site wastewater disposal. However, it is important to keep in mind that there are a variety of other pollutants that may be present in domestic wastewater, many of which are not regulated under either State ground water or drinking water standards. Detection of these compounds in well samples thus serves as an indicator of the impact of on-site systems on ground water quality. Some of the most persuasive evidence of contamination from on-site disposal systems in the East Mountain area are the results of the USGS analyses for organic compounds. Although these constituents were detected at trace levels, detection of multiple contaminants in each sample from the East Mountain monitoring wells supports the finding that there is in fact changes in ground water quality due to the use of on-site disposal systems. This information is particularly valuable in

light of the absence of excessive nitrate concentrations, possibly due to the presence of reducing conditions in the area. It is recommended that the monitoring well network established during this study be re-sampled on a bi-annual basis and analyses for anthropogenic organic compounds be repeated. Results of these analyses will provide confirmation of the presence of these contaminants in underlying ground water. Continued monitoring for these constituents will allow determination of trends of contamination.

1.18.4. Isotopic Analysis of Ground Water Samples

Much of the interpretation of the results of the Sandia Heights ground water samples is based on the model prediction that the concentration front of the nitrate contaminant plume has not yet reached the down gradient monitoring wells. This interpretation can be tested, at least in principle, by dating the ground water using isotopic analyses. Common isotopes used to date water include hydrogen (deuterium (^{2}H) and tritium (^{3}H)) and oxygen (^{18}O and ^{16}O). In addition, isotopes of nitrogen (^{15}N and ^{14}N) have been used to determine the source of nitrogen compounds in water samples. Isotopic analysis of other ground water samples in the Albuquerque Basin has been performed previously. Comparison of the isotopic characteristics of the water samples from the areas studied in this project with water samples from the City and County deep nested piezometer network would have the added benefit of providing additional data on the isotopic variability within the Basin and could be used to determine the nature of the hydraulic connection between shallow and deep ground waters.

1.18.5. Investigate Nitrate Transport and Transformations in Thick Unsaturated Zone

The vadose zone investigation conducted for this study was very preliminary and limited in nature; the level of effort represented less than 10% of the total amount of work on this project. Some of these limitations included: 1) soils were only sampled to 65 feet which represented about 10% of the total saturated thickness, 2) the site investigated was relatively new, and, 3) pore water chemical analyses were limited to analyses of leachate from core samples. Despite these limitations, the results were very provocative. These included: 1) the finding of

very rapid flow of water through the soil, 2) rapid nitrification in the soil, and 3) the presence of high concentrations of organic carbon (represented by chemical oxygen demand or COD) in soil pore water at the 65 foot depth. The results may also show a decrease of nitrates with depth throughout the entire 65 foot depth, which suggests that microbial denitrification is occurring in the soil column. However, the well aerated nature of the soil should result in aerobic conditions through out the unsaturated zone that would prevent denitrification from occurring. One possibility is that anoxic microenvironments are present in which denitrification may take place. Further investigation is needed to determine if there is in fact a reduction of nitrate concentration with depth, and whether microbial denitrification by heterotrophic organisms are responsible. The field study should include investigations at sites on both sides of the Sandia Mountains.

In order to answer these two questions a deep core sampling program is needed and it should be complimented by installation of appropriate equipment to allow monitoring of soil redox conditions with time. Recent experience by UNM personnel with a sonic drill rig at a UST site has demonstrated the ability of this type of drilling equipment to rapidly drill to depths in excess of 200 feet while collecting near continuous samples of undisturbed core. This core sample could be used to repeat the testing procedure done for this study on samples from much greater depths. Recovery of more core material would also enable conducting a wider variety of analyses. The second part of a deep coring program should include installation of monitoring equipment to measure subsurface properties that are relevant to transport and transformations of nitrogen species. Examples might include installation of suction lysimeters, neutron access tubes for measurement of soil moisture content, gas sampling tubes, and dialysis bags for collecting soil water and gas samples. These devices would allow measurement of soil gas oxygen levels, as well as in-situ collection of soil water samples for chemical analysis. As with the study described in this report, consideration should be given to use of tracers to facilitate collection of data regarding the vertical migration rate of soil water flow through unsaturated soil.

A field research program similar to that described here would likely be quite expensive. Furthermore, it is recognized that neither the City of Albuquerque nor Bernalillo County are entities which have limited resources for funding basic environmental research. Accordingly, it is recommended that the City and County seek opportunities to collaborate with more traditional Federal or State research funding agencies and leverage their resources to support a deep vadose

zone contaminant transport and transformation investigation. One of the selling points that can be made is that work performed to date by the City, County and their contractors has established a large base of knowledge and experience, as well as a suite of ground water monitoring wells that can result in a more productive investigation than if the study was conducted elsewhere.

1.18.6. Develop a Geographic Information System for Ground Water Quality in Bernalillo County

The information gathered from the monitoring well network developed in this study should be incorporated with that from other City and County wells, including both production wells and monitoring wells. This type of information seems appropriate for inclusion in a GIS data base. It is thus recommended that the information be included in the Albuquerque GIS, and that the results be made available for use by water management professionals.

1.18.7. Collect Data Relevant to On-Site Disposal Systems During Other Ground Water Quality Investigations

Construction of a ground water quality monitoring network is very difficult, extremely expensive, and in many cases limited by drilling site access. For example, in this study nearly half of the cost of this project was associated with drilling contracts, and approximately half of the time spent by project staff was devoted to locating and installing the monitoring well network. There are numerous other Federal, State and local programs which also involve installation of ground water monitoring wells. These include investigations conducted under federal RCRA and CERCLA (Superfund) regulations, and the State leaking petroleum underground storage tank (UST) program. Most of these investigations are directed towards determining the extent of contamination of specific pollutants such as petroleum hydrocarbons or chlorinated solvents and chemical analyses of ground water samples are limited to these target pollutants. Monitoring wells at two UST sites in the East Mountain area were sampled during this study and found to have high nitrate concentrations. These two UST sites were not sufficiently close to high density subdivisions to be included in the ground water monitoring

program, however, they represent a very low cost source of additional information that may be of some value in assessing the impacts of on-site disposal systems on ground water quality.

It is recommended that the City and County adopt a policy to encourage analyzing ground water samples from monitoring wells for major ion chemistry (i.e. cations and anions), major nitrogen species (organic N, ammonium, and nitrates), and chemical species which indicate the redox conditions of the water sample (i.e. dissolved oxygen, iron, sulfides, and manganese). This policy would request that these analyses be performed at least once at all monitoring wells at a site. Furthermore, the City and County should be allowed the opportunity to re-sample the wells for these parameters if future results would help understand subsurface phenomena. Hydrologic information such as depth of the water table and the piezometric surface, as well as the water quality data would be reported and entered into the ground water GIS data base. With time it is anticipated that the data accumulated by establishment of this policy will allow City and County water managers to develop a much better understanding of the quality of the regional ground water resource, and to detect changes in this quality as a result of land use and development at the surface.

1.19.   Final Thoughts

Perhaps the most significant conclusion of this investigation is a reaffirmation of the diversity and extraordinary complexity of the hydrogeologic environment which occurs in Bernalillo County. Due to this variability and complexity it is important to recognize that the types of ground water problems resulting from on-site wastewater treatment and disposal will also vary. Therefore, it is recommended that local and state ground water protection regulations should not focus on a single water quality criteria, but rather they should address the protection of the overall quality of the ground water resource. In particular, past regulatory practice has been directed primarily towards prevention of nitrate contamination of ground water. This is due in large part to the fact that high nitrate concentrations can often be unequivocally attributed to on-site disposal systems, and because they are cheap and easy to measure. In contrast, measurement of synthetic organic compounds in a ground water sample requires special sampling and analytical procedures that are difficult and very costly. This study has demonstrated that subsurface environmental conditions in many areas of Bernalillo County are

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not suitable for generation of nitrates, yet that doesn't mean that ground water contamination has not occurred or will not occur in the future.

In spite of the limitations associated with this investigation, it has shown that on-site disposal of wastewater has resulted in ground water contamination and changes in ground water quality in each of the four areas studied. This contamination varies in nature and degree, and includes elevated nitrate concentrations beneath the Sandia Heights and Sandia Park subdivisions, and detection of synthetic organic compounds in all of the East Mountain subdivisions studied. Future land use planning policy must recognize this problem.

Continued monitoring of ground water quality in the County is imperative, particularly in light of model predictions which suggest further deterioration of water quality. This monitoring should include sampling and analysis for a spectrum of contaminants including both inorganics, organics, and indicators of microbial quality.

Finally, in light of the findings of this study that ground water contamination has occurred in a variety of hydrogeologic environments throughout the County, development and implementation of stringent ground water quality protection is needed. One of the original goals of this study was to assist the City of Albuquerque and Bernalillo County in developing land use criteria to minimize ground water quality impacts of future developments in unsewered parts of the County. These criteria might include constraints on the density of future housing developments, requiring us of on-site treatment systems which provide a higher degree of wastewater treatment prior to discharge, and correlation of site hydrogeologic characteristics to the type of on-site treatment system which might be allowed. The complexity identified by this investigation suggest that no single constraint will provide adequate protection. Furthermore, at many locations in the County it is not possible to quantitatively predict the impact of on-site disposal systems on underlying ground water quality. Therefore, development of aquifer protection regulations will require a high degree of technical judgement by City and County staff. These regulations should also include a degree of conservatism to allow for the variability and uncertainty associated with the subsurface environment.

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